A study to examine the benefits of the End of Life Vehicles Directive and the costs and benefits of a revision of the 2015 targets for recycling, re-use and recovery under the ELV Directive

Final Report to DG Environment

In the framework of the contract to provide economic analysis in the context of environmental policies and of sustainable development

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PART A: INTRODUCTION AND CONCLUSIONS

1 INTRODUCTION

1.1 Background

Disposal of cars and light commercial vehicles at the end of their operational lives (end of life vehicles – ELVs) is estimated to generate over 10 million tonnes of material requiring treatment and disposal in 2005 within the EU. This volume is projected to increase to 14 million tonnes by 2015 as the number and average weight of vehicles increases.

The problems created by the generation of ELV waste arisings have been addressed by Directive 2000/53/EC (the 'ELV Directive'); which aims to reduce the amount of hazardous waste, increase the re-use, recycling and recovery of materials from ELVs and to improve the environmental performance of operators involved in the production and maintenance of vehicles and in the treatment of ELVs.

1.1.1 Overview of the Directive

The Directive contains 13 articles which outline the objectives, definitions and scope of the Directive (Articles 1-3), establish requirements for waste prevention (Article 4), set requirements for the collection of ELVs (Article 5), set environmental standards for treatment (Article 6), establish targets for reuse and recovery (Article 7), and specify coding standards and the provision of dismantling information (Article 8). The remaining articles deal with reporting, implementation and administrative procedures. Annex I specifies in more detail the minimum technical standards relating to treatment in accordance with Articles 6(1) and 6(3), while Annex II lists certain materials and components exempt from the restrictions relating to hazardous substances in Article 4 (2)(a).

1.1.2 General Approach to ELV Treatment

The general technical process of dealing with ELVs is presented below (Figure 1.1):

Figure 1.2 provides a more detailed overview of how ELVs arise and are treated in the EU. Vehicles reach the end of their life either because they become old and worn out and cease to be roadworthy ("natural" end of life vehicles or NELVs) or because they are written off following involvement in an accident ("premature" end of life vehicles or PELVs). Vehicles may be sold for export either before reaching the end of their life as second hand vehicles, or at the point of deregistration as waste. Conditions for sales of second hand vehicles and ELVs (waste) are different since only the latter ones would be subject to waste legislation.

Figure 1.1: Overview of the ELV Treatment Process

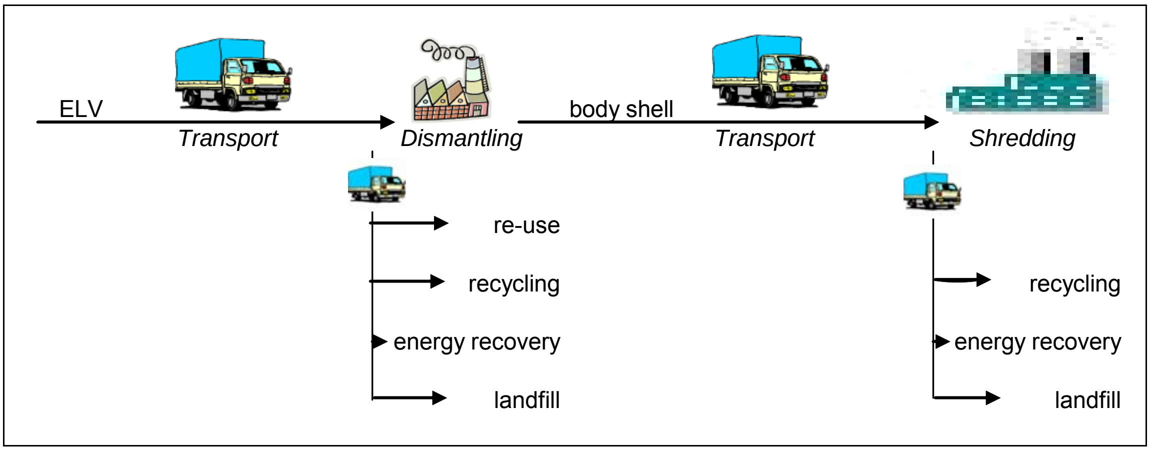
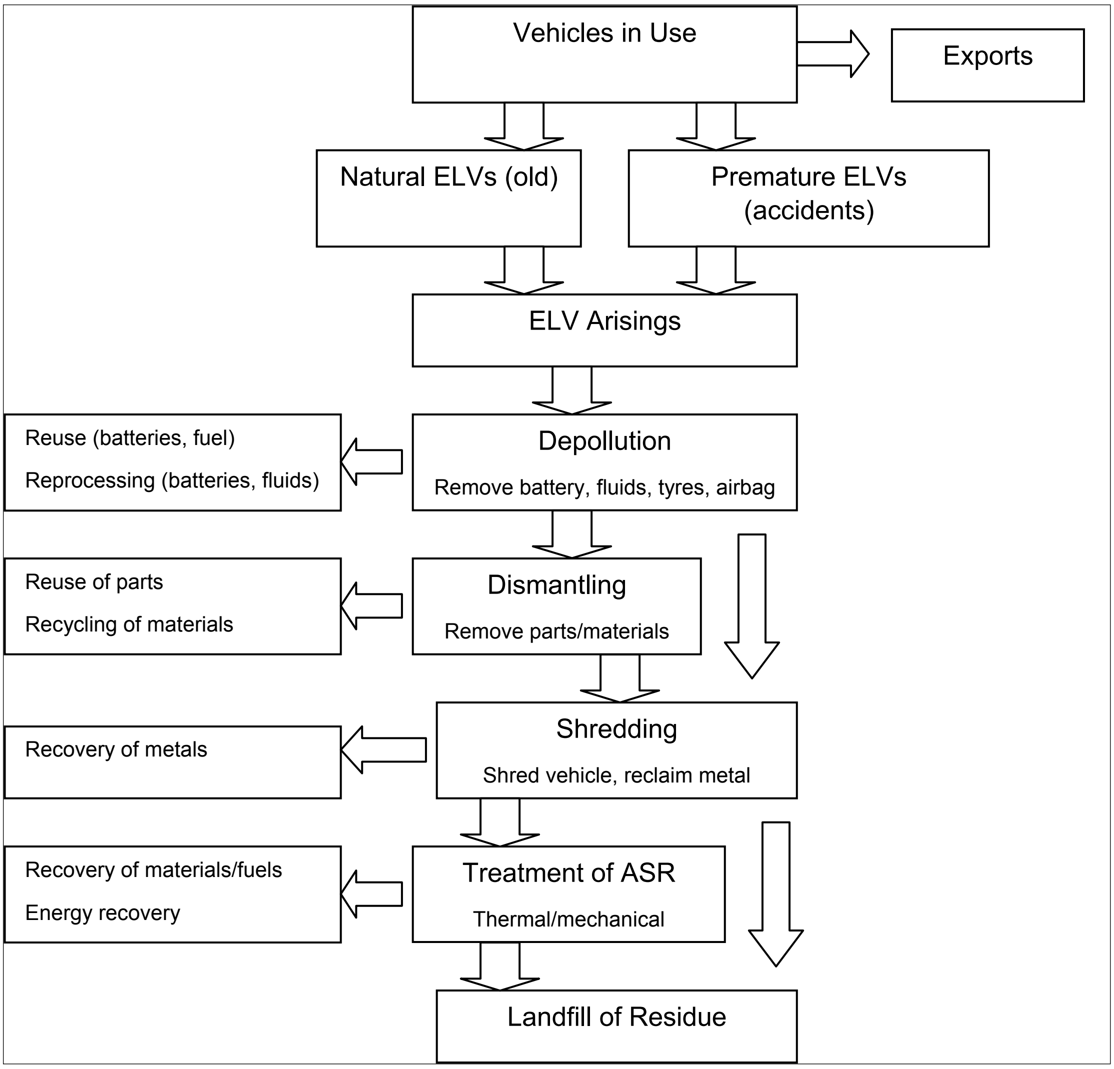


Figure 1.2: Description of ELV Arisings and Treatment



Those vehicles that are not exported need to be treated at ATFs and undergo a process of depollution, involving the removal of fuel, oil and other liquids, as well as the battery, airbags and heavy metals. Some ELVs may be dismantled to remove valuable parts and materials, before they are sent to a shredder; others will be sent directly to the shredder. Shredding involves a capital intensive mechanical process and results in the recovery of metals from the vehicle, leaving auto-shredder residue (ASR), a combination of materials such as plastics, textiles and glass. ASR has traditionally been disposed of in landfill sites but is increasingly being treated to separate useable fractions and enhance rates of recycling.

1.1.3 Target Rates of Treatment of ELVs

From 1 January 2006, 85% of ELV material by weight must go for recovery, re-use and recycling, with at least 80% of ELV material by average ELV weight going for recycling and re-use.

The ELV Directive requires economic operators to meet the cost of achieving these targets through the requirement to ensure free take back of ELVs. In other words, any costs associated with meeting the targets above the value of an ELV at the time of de-registration can not be charged to the last user.

To the extent that costs are incurred by public authorities or car producers above the value of ELVs, it may be that some form of additional charge to cover these costs in part or in full could be levied on the sale of new cars. The first example in the EU of this practice is in the Netherlands where, as part of a complete system for auto recovery (ARN), a waste disposal fee (45 euro) is levied on the first registration of the vehicle, which part finances a network of registered recycling and dismantling operators, and which meets the ELV targets for 2006.

Under the current provisions of the Directive, the proportions of ELV material required to go for recovery, recycling and re-use are due to change in 2015. Targets for 2015 onwards were set at 95% for re-use, recycling and recovery, and at least 85% recycling and re-use.

Table 1.1: Current Target Recycling and Recovery Rates for ELVs

Target Dates	Recycling & Re-use	Total Recovery, Recycling & Re-Use
2006	80%	85%
2015	85%	95%

Note: % by average weight of ELV

The higher recycling and recovery targets reduce the amount of waste that can be disposed of to landfill (the typical disposal route). The amount of waste that has to be treated depends on the actual weight of an ELV, which is estimated to be on average 1,025 kg in 2015, compared to an average weight in 2006 of 964 kg.

1.2 Objectives of this Study

Articles 7(2) and 7(3) of the Directive require that the 2015 recycling, re-use and recovery targets set in the Directive be reconsidered by the European Parliament and the Council. To this end, the European Commission is required to present a report, accompanied by a proposal, to inform discussion of these targets. In its report, the Commission shall take into account the development of the material composition of vehicles and any other relevant environmental aspects related to vehicles. In line with the objectives of better regulation, the Commission would like the decision on these targets to be based on evidence of the environmental and economic costs and benefits of their achievement.

This study has been prepared to provide the Commission with that information and therefore:

> A: Reports on the economic and environmental benefits arising from the ELV Directive to date.
>
> B: Appraises the costs and benefits of a range of potential targets for re-use, recycling and recovery of ELVs from 2015.

1.3 Methodology and Process

1.3.1 Baselines

Both Task A and Task B require the collection and analysis of data on the current situation with ELV treatment – for Task A to establish the economic and environmental benefits of the Directive to-date, reviewing the benefits to operators (vehicle producers and the ELV treatment industry), government and society generally. For Task B, this data is needed to establish a suitable baseline against which the impacts of future targets can be judged.

1.3.2 Task A Baseline

The general approach has been to consider the benefits achieved to date, by reference to the treatment of ELVs prior to the MS and EC initiatives which led to the Directive in 2000.

No systematic data exists of this situation; for example, there was no formal ex ante assessment at EU level of ELV initiatives and which could be considered to provide a reference point. Moreover, the reporting obligation on the implementation of the ELV Directive (Art. 9 of the Directive) has not begun in time to provide data.

In light of the above, the approach has been to adopt a ‘market led’ description, i.e. to assume that the treatment of ELVs prior to public intervention was determined by the purely commercial costs and benefits associated with treatment. This in turn means that the initial reference point is one where ELVs were scrapped in order to obtain and sell the scrap metal for commercial return. In addition, a number of parts were reconditioned and reused, including batteries, with a proportion of used tyres sold for re-treading.

Since current implementation of the Directive varies between Member States, the current levels of benefits will vary. The work has sought to examine progress in selected Member States to represent varying levels of implementation to illustrate the order of magnitude of benefits to-date. The experience of the Netherlands is of particular value in this context, given that it is widely appreciated to have implemented the provisions of the Directive, more fully than other MS, at the present time.

1.3.3 Task B Baseline

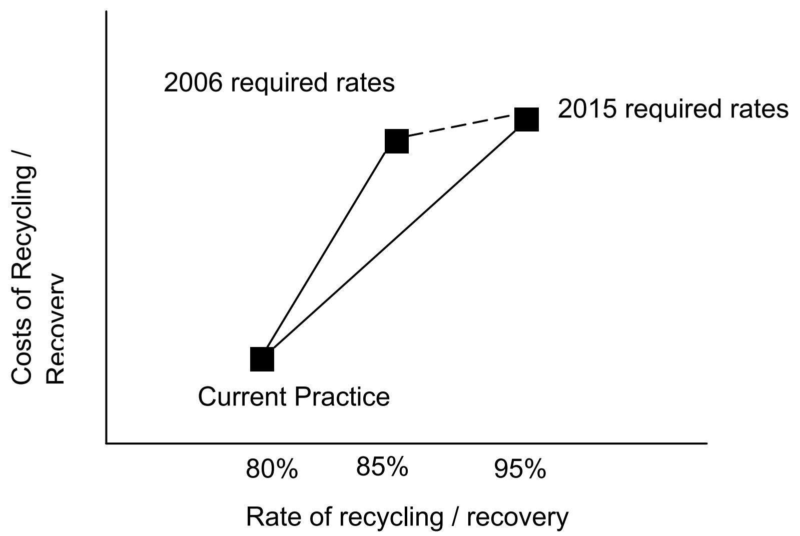
The estimation of the additional costs and benefits of moving from the 2006 to 2015 targets requires appraisal of the technical options available. Different MS are at different stages of implementation of the 2006 targets. For example, in the Netherlands, the additional cost might only relate to the planned change in targets between 2006 and 2015. In the case of other MS which have yet to meet the 2006 targets, the additional costs and benefits relate to both the introduction of the 2006 and 2015 targets.

Consequently, to ensure consistency and a transparent baseline, the baseline for the estimation of the additional costs and benefits is first taken as the 2006 target level of re-use, recycling and recovery, rather than the actual rates expected to be achieved in 2006.

However, to indicate the change in costs as higher rates of re-use, recycling and recovery are achieved from different starting points, we have also separately estimated the costs of meeting the 2006 targets and the 2015 targets compared to a baseline defined by the current practice 'market-led' approach to ELV recovery but taking into account the depollution provisions of the ELV Directive. This approach has the advantage of not having to assume that the technical responses to achieve the 2006 targets are those developed in countries such as the NL, but rather allows MS to develop the most cost-effective response to 2015 targets. This comparison also takes into account (given an average vehicle life of 13 years) changes in vehicle design leading to changes in composition and weight of ELVs between the early 1990s (and treated in 2006) and early 2000s (and treated in 2015).

The difference between the costs for 2006 and for 2015 (the dashed line in Figure 1.3) represents the additional cost of the potential 2015 targets. Note the diagram is only illustrative and does not indicate the actual costs.

Figure 1.3: Indicative Cost Assessment



1.3.4 *Appraisal of Potential of 2015 Targets*

Task B was to examine the costs and benefits of a range of potential targets for re-use, recycling and recovery, to start in 2015. To tackle the task, the study examined different technical options for achieving higher targets than the 2006 targets. In particular, this involved researching the inputs and outputs of different processes for dismantling, mechanical recycling and recovery.

During this work, it became apparent that use of Post Shredder Technologies (PSTs) to further process the residue coming out of shredders offered the most cost-effective route for achieving higher targets.

This research also indicated the maximum likely recycling and recovery rates achievable by PSTs and the additional costs of higher 2015 targets. As explained later, the cost structures of PSTs suggested that the same technologies appeared likely to be used for achievement of any of the range of targets, even the 2006 targets. The presentation of this study therefore is not set out as a repetitive description of the costs and benefits of different targets (e.g., ways to meet 90% recovery compared to 95% recovery), but instead looks, initially, at the costs and benefits of using PSTs operating at their potential.

Therefore, considering an average composition of ELV in 2015, the work has determined different technical options (and combinations of options) to reach the specified targets. These different options and combinations allow a small number of scenarios to be defined.

Each scenario is characterised by a particular use of techniques for re-use / recycling / recovery defined in relation to each of the component / materials comprising an ELV. (This requires assumptions about the average total weight and composition of an ELV and the average weight of each component / material fraction). For each scenario the additional costs have been identified (compared to the 2006 targets); together with the wider economic and environmental costs and benefits in order to assess whether potential targets for 2015 represent net societal benefit.

The analysis of environmental benefits was directed to establishing the relative environmental benefits of, in the first place, the various treatment options compared to landfill, and, secondly, of the recovery options against mechanical recycling. The assessment of different scenarios was based on an estimate of the change (increase or decrease) in the volume of materials per ELV treated between 2006 and 2015 in different scenarios, including landfill.

The assessment was based on available LCA analyses for the different materials found in ELVs and especially for different plastic vehicle components. The assessment is constrained by the absence of available data on all plastic resins and only a limited coverage of other materials.

Attempts to quantify the relative costs and benefits of treatment options have extended to a consideration of the externality value attached to 5 impact categories (air acidification, greenhouse effect, photochemical oxidation, eutrophication, disamenity) as measured for each of the different treatment methods. Although there are many uncertainties (some directly linked to the monetisation methods themselves and others occurring when combining results from monetisation and LCA), it provides an indication of the order of magnitude of the environmental benefits. It results from the calculations that external costs have a profile similar to greenhouse effects. It should be noted that, within the framework of a multi-criteria analysis such as LCA, externality quantification is a method to aggregate the different environmental impacts which are quantified. Other aggregation methods exist, which may give different relative weights to the various impact categories and may thus result in different conclusions.

1.3.5 Method of Approach

The study was conducted using a combination of national case studies, consultations with stakeholders as well as a series of interviews and discussions with waste operators and related technical specialists, and supported by desk-based research.

In the first part of the study case studies were carried out for the following countries:

- France
- Germany
- Netherlands
- UK
- Hungary
- Poland
- Spain
- Australia

The national case studies looked at the current management of ELVs including the extent to which current targets were likely to be reached, examined the technology options for re-use, recycling and recovery, reviewed the vehicle and recycling markets and assessed current environmental benefits. Case-study information was obtained through internet searches, reviews of the literature, as well as telephone interviews with relevant organisations and stakeholders.

The case study assessments provided some evidence that achievement of the 2015 targets would have to be met through increased recycling of ASR (Auto Shredder Residue) resulting from the shredding process. An international review of the current status and future potential of post-shredder technologies was therefore undertaken. Several different technologies were identified. For each one, with the assistance of the relevant business, an approximate 'mass-balance flow' was estimated to understand the use of treatment of post shredder residues and the consequent production of materials for sale and the generation of associated waste streams. In most cases there was an extensive dialogue with both the engineers and/or management staff of the technology companies to ensure that the information presented was correct. Previous studies examining these technologies, particularly costs, were also referenced.

To inform then research and to assist in checking preliminary conclusions, telephone interviews were carried out with a range of stakeholders, including car manufacturing associations, material recycling associations, and environmental institutes.

2 CONCLUSIONS AND RECOMMENDATIONS

2.1 Current Benefits of the ELV Directive

2.1.1 Summary of Conclusions

In summary we find that:

- There are significant environment benefits already being realised from the Directive
- These are due to the standards that are set for authorised treatment facilities (ATFs) for the depollution, dismantling and treatment of ELVs
- There are also significant economic benefits from the Directive
- Generally neither the environmental or economic benefits are fully realised because of the lack of full implementation and enforcement of the Directive
- Compliance with the Directive's 75% recycling target does not bring any additional environmental benefit in most member states (MS) because recycling would take place at this level due to the economic incentives currently provided by market forces, in particular the market prices for scrap metal.

The estimates of current benefits are based on data available from MS where there has been compliance, unless otherwise stated.

2.1.2 Varied Levels of MS Compliance

The review of activity in a set of national case studies demonstrates a varied level of compliance with the Directive in Member States. This depends largely on the systems in place prior to the implementation of the Directive. Those MS with systems for collection and treatment of ELVs of longer standing (e.g. NL, Dk, SE) have been able to ensure that nearly all treatment facilities are authorised. In other countries (such as France and the UK) some authorisation has taken place, but not under the terms of the Directive and with little systematic monitoring to establish the number of authorised facilities. As a result, even in France it is estimated that perhaps as many as 40% of treatment facilities are operating illegally without authorisation. In Hungary (and by extension other eastern European MS) the rate of illegal operation might be as high as 80% of all treatment facilities. As a consequence, the benefits to-date vary between Member States and are significantly smaller than would be the case with a fuller level of compliance.

2.1.3 Direct Economic Benefits

Direct economic benefits from the Directive are non-trivial. They include:

- The promotion of resource efficiency by providing incentives for innovation in both vehicle design and the treatment of ELVs. There is limited evidence of any significant change in vehicle design at present as a result of the Directive;

however, vehicles manufacturers are currently legally bound to design their vehicles in such a way that they meet the recovery and recycling targets set out in the ELV Directive.[1] There is significant evidence that the Directive is technology forcing in the treatment sector, with the development of technologies capable of reducing treatment costs over current methods.

- The improved efficiency of the treatment sector, with the case studies reporting an improving technical and professional approach, with investment leading to higher operating efficiency, securing its longer term future. These benefits are still modest compared to what might be expected with full compliance, leading to overall lower operating costs.

- The reduction in direct waste disposal costs associated with the avoided landfilling of ASR. These direct costs are wholly or partly offset by other treatment costs, but lead, at the margin to lower landfill costs to other waste producers.

- The reduction in levels of abandonment of vehicles and hence the cost to the public sector of having to collect, store and organise disposal. Quantifying the benefit is difficult because levels of abandonment are partly determined by the value of ELVs which in turn are significantly influenced by scrap metal prices.

- The reduction in vehicle crime attributable to the requirement to issue CoDs, improving the effectiveness of the vehicle registration system and reducing the scope for running vehicles illegally, for using unregistered vehicles to undertake crime, and for abandoning vehicles with impunity. The UK's Regulatory Impact Assessment estimated that the Directive itself would yield annualised benefits valued at £21 million (30 million euro) between 2007 and 2025 as part of broader package of measures.

2.1.4 Current Environmental Benefits

The varying levels of compliance also affect the level of environmental benefits achieved to-date. A detailed analysis linked to the depollution requirements of the Directive has been undertaken, which attempts to take into account the considerable overlap with other policy measures influencing, for example, the treatment of tyres and batteries.

The assessment, based on a wide range of consultations with relevant stakeholders, indicates in particular that the directive is likely to be responsible for:

- An increase in the number of vehicles treated in authorised treatment facilities (see Topic 1 in Section 4)

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[1] This obligation, introduced by Directive 2005/64/EC on the type-approval of motor vehicles with regard to their reusability, recyclability and recoverability aims to ensure that all type-approved vehicles belonging to category M1 and N1 may be put on the market only if they are reusable and/or recyclable to a minimum of 85% by mass and are reusable and/or recoverable to a minimum of 95% by mass. Directive 2005/64/EC of 26 October 2005 on the type-approval of motor vehicles with regard to their reusability, recyclability and recoverability, OJ. L. 310, 25.11.2005, p. 10.

- An increase in the operating standards of treatment facilities even where not in full compliance with the ELV Directive yet (see Topic 1, Section 4).

- An increased level of treatment for different materials.

- Specific environment improvements and related health improvements as a result of:

  – A reduction of over 50,000 tonnes of waste oils and other fluids

  – Energy savings from the regeneration of waste oils

  – A reduction in disposal of up to 4,000 tonnes of sulphuric acid from batteries and a similar amount of lead, with up to three quarters of a million batteries safely recycled

  – A reduction in the volume of tyres and glass disposed of to landfill

  – Improved management of liquid gas tanks

- Continuing improvements in environmental quality as the Directive is more fully implemented.

- An increased number of technical feasibility studies into the recycling possibilities for plastics which is likely to increase the benefits and scope of plastics recycling in the future, moving the EU towards its goal of being a 'recycling society'.

2.2 Costs and Benefits of Potential 2015 Targets

FINDINGS

Scene Setting

- The volume of ELVs requiring treatment in the EU25 is increasing. The average weight of ELVs is increasing. Combined these trends indicate an increase in arisings requiring treatment by 2015 from approximately 10 million tonnes to 14 million tonnes.

- Although there is a significant trade in second hand vehicles, with net exports to lower income MS, five MS (Germany, UK, France, Spain and Italy) are responsible for approximately 75% of EU 25 vehicle deregistrations.

- The current levels of reuse, recycling and recovery vary significantly between MS, with the highest rates being achieved in the Netherlands with a reuse and recycling rate of approximately 85%. The scrap value of parts and metals is sufficient to ensure that left to the market at least 75% of the ELV is recycled. With some efforts to increase recycling and the requirement for depollution current practice achieves approximately 80% reuse and recycling – a level which can be achieved without use of new techniques.

- Currently, the remainder of the ELV material, after post shredding processes designed to recover the metal, is landfilled as Auto Shredder Residue (ASR).

There is very little use of incineration as a means of disposal, partly because of the very limited capacity in countries like the UK and Germany.

- Changes already made in vehicle design will change the nature of ELV arisings in 2015 compared to 2006 in terms both of weight and composition. The average weight of ELVs will increase, and the share of an ELV by weight accounted for by plastics will increase while the share accounted for by ferrous metals (mainly steel) will decline. However, the absolute weight of metals will increase.

- Table 2.1 summarises the amount of material that has to be treated to reach higher rates of recycling and recovery, taking into account the increase in ELV weight. In 2006 80% by weight of ELV is recycled and 85% recycled or recovered. The required increase in the recycling rate (2006 to 2015) from 80% to 85% is equivalent to recycling an additional 39 kg (193kg – 154kg). The increase in the overall recycling/recovery rate (from 85% to 95%) is equivalent to treating an additional 93 kg (145kg – 51kg).

Table 2.1: Indicative Levels of Treatment Required Under Different Targets

	Allowable Levels of Landfill (kg)			
	Recycling / Recovery Rates			
Average ELV Weight (kg)	80%	85%	90%	95%
954 in Base	189			
964 in 2006	193	145	96	48
1,025 in 2015	205	154	103	51
Diversion Required with Higher Targets (kg) - 2006 to 2015				
Recycling (from 80%)		-39	-90	-142
Recycling & Recovery (from 85%)			-42	-93

- This review indicates that the compliance with the 2015 targets compared to 2006 targets requires diversion and treatment of up to 100kg of material per ELV.

- Until recently the approach to increasing rates of recycling has assumed to be in the form of higher rates of dismantling, with the remaining material capable of incineration and providing energy. However, the more specialised option of post shredder treatment (PST) using new technology has been developed, driven in part by the more stringent requirements for ELV treatment in Japan and Switzerland. This technology is being developed by a number of different operators and uses different techniques; mechanical separation methods to recover and recycle material or thermal treatment which processes materials as energy feedstocks.

- The targets in the Directive, with high marginal costs of dismantling beyond the next say 30kg to 50kg of material (or 3% to 5% by weight), the increasing real cost of landfill and the lack of incineration capacity, has seen the setting up and piloting of PSTs in Germany and France. A major new investment has recently been announced using mechanical separation designed to treat ELVs in the Netherlands and Germany.

- PSTs also have a role to play in meeting the requirements of the WEEE Directive. Approximately half of the shredder residue treated is derived from ELVS. The remainder is largely WEEE material. Therefore technological advances stimulated by the ELV Directive will bring benefits to the economy and environment beyond the direct benefits of ELV treatment.

Technical Feasibility

A review of current and emerging technical solutions, suggests that whilst dismantling maybe technically feasible, the costs of extending rates much beyond current levels rise extremely steeply because of the increasing labour input per kg treated.

Partly as a consequence of this, there is increasing investment and development of technologies to treat Auto Shredder Residue (ASR), in Post-Shredder Technologies (PSTs) either using mechanical separation for recycling or thermal treatment based on energy recovery; although thermal treatment plant also include an element of mechanical recycling, mainly of metals.

The review indicates that conservatively these PSTs are able to achieve recycling or recovery efficiencies of at least 75% (ie a maximum of a quarter of the material treated subsequently requires disposal). Given baseline rates of recycling and recovery of around 80% per ELV (including depollution), these technologies are capable of treating the remaining 20% and enabling an overall recycling/recovery rate of 95%.

Attributing Costs to Higher Recycling / Recovery Targets

In most MS the achievement of the 2006 target of an overall rate of reuse, recycling and recovery of 85% has yet to be reached. The high costs of dismantling and the emergence of PSTs means that it is likely MS will seek to use PSTs to meet *both* the 2006 *and* 2015 targets.

If this is the case then there is an argument that, since the investment and operational costs will have been incurred to meet the 2006 target and, since plants are capable of achieving the 2015 targets, there are no additional costs of meeting higher targets. The PSTs are characterised by high fixed costs, which makes it commercially sensible to operate the plants at full capacity, achieving the maximum levels of treatment. As a result the plant operating in the counterfactual situation (ie without higher targets) is the same as the plant operating with higher targets. Since there is no change to the nature of the plant required by the higher targets, there is no additional cost attributable to the higher targets. The costs of the PSTs are set against the achievement of the 2006 targets.

However, it could also be argued that the higher targets are the rationale for the investment in PSTs, and that in the absence of the higher targets the PST option would not be pursued and MS would seek to use existing techniques to meet the 2006 target.

The main cost analysis presented in the report provides the ‘ceiling’ costs associated with the higher targets, attributing the full costs of treatment above 2006 levels to the 2015 target. This compares with zero as the ‘floor’ costs assuming that 2006 targets are met by the same plant operating at levels that also deliver the 2015 targets.

Costs based on variable costs provides an intermediate estimate. This approach seeks to recognise that there are costs of higher levels of treatment, which could be attributable to the higher targets, but that since investment was made to reach the 2006 targets the fixed costs are not attributable to meeting higher targets.

Treatment Costs

The review of technical options suggests that on a per kg basis, dismantling is the most expensive option, post shredder mechanical separation and recycling the cheapest option, with the costs of post shredder thermal treatment recovery options more costly than mechanical separation, but significantly less costly than dismantling.

Operators of the treatment options will charge waste producers a 'gate fee' based on the difference between the processing costs and recycling / recovery revenues (which are in turn dependent on the markets for recycled or recovered material).

It is typically assumed that additional treatment of waste materials must imply an additional cost. However, this is to assume that in all cases the gate fees are greater than the avoided costs of landfill which would otherwise have to be incurred. The identified costs of treatment indicate that in MS with high landfill costs it is possible that the diversion of material from landfill is capable of yielding cost savings to the waste producer.

Review of the costs of the emerging PST processes reveals them to be subject to significant economies of scale. The range of unit costs used in the analysis reflect this effect with higher unit costs associated with smaller plants (of say less than 100 ktonnes of capacity) and medium and lower unit costs more representative of the costs of larger plants (of around 200 ktonnes or larger).

Technology Forcing

Evidence from Japan and Switzerland is that higher targets have forced technological development, through new alliances between vehicle producers and waste treatment operators – and that this is now happening in the EU in higher cost MS, with new investment in both mechanical and thermal treatments. Without higher targets the pressure to develop these technologies will be reduced and the benefits of cheaper and potentially more environmentally effective treatment methods forgone.

The closer collaboration with the treatment sector is likely to increase the opportunity for innovation in design to reduce ELV treatment costs. For example, LCA analysis indicates the sensitivity of environmental benefits to the types of resins used in vehicles. Further work should indicate which are the more beneficial resins and the scope to substitute these for less beneficial ones.

Economic Efficiency and Cost Savings

There are a number of potential economic benefits as a result of higher targets and the associated introduction of PSTs:

- The work indicates that if landfill costs are high the new technologies could improve economic efficiency by providing a saving in costs to waste producers. This is determined by the treatment plant efficiency, which has important scale economies, and by future prices for recyclates and for energy as well as future landfill costs. In this context the PST is likely to take all of the material from auto shredders rather than adopt a piecemeal approach to taking individual fractions.

- Treatment processes require energy, and rising real energy costs will increase treatment costs. However, in the case of recovery, real increases in energy costs over the next 10 years are likely to substantially reduce the gate fees for treatment beyond those currently identified. The key recyclate price is for granulated plastics, which is also likely to be strongly influenced by energy prices. Where real energy costs increase significantly, the use of PSTs are likely to improve economic efficiency.

- The development of more advanced recycling technology will enable greater recycling of metal fractions of ELVs than at present. Metal prices are likely to remain high as world economic growth continues, making this a significant advantage.

- Increased recycling and recovery will reduce the EU's dependency on imports of energy and materials, improving the EU's trade balance. Greater resource efficiency will aid EU competitiveness.

- Evidence is emerging from a number of MS (eg UK, Germany) of a significant shortage in industrial and municipal waste treatment capacity over the next decade. This is likely to mean that even in the absence of the Directive, new investment in treatment capacity (including for ASR) would be needed. Shortage of treatment capacity also exists in the cohesion and accession countries as evidenced by the need for structural fund allocations. This investment requirement is likely to mean real increases in landfill and incineration costs, especially where minimal or non-compliant plant is replaced, and where stringent conditions are applied in order to enable these developments to proceed. This means that cost savings from PSTs are more likely; and that the treatment sector will secure greater returns to investment in PST rather than conventional treatment capacity. It also means that there is little or no sunk cost in existing treatment capacity that might drive reductions in landfill prices. Note that the cost scenarios do not provide for possible sunk costs.

Competitiveness

- Review of the impact on European competitiveness through the effect on European vehicle producers suggests that this will be limited, with potentially some benefit to those producers with significant European market share able to spread the fixed costs of organising the necessary take back schemes.

- The review of the impact of higher targets on the treatment sector has been based on a consideration of the possible increase in capacity associated with the introduction of post shredder technologies. The vertical integration of existing shredder operations, which are significant capital intensive businesses, to incorporate the new technology would suggest the rationalisation of the waste treatment sector into fewer larger plants with less reliance on smaller dismantling and recycling operations. The capital intensity of the operation relative to landfill operations suggests that the additional capacity is unlikely to generate many if any additional jobs. However, the investment in technology development is likely to make the sector a stronger competitor globally, with potentially longer term benefits to EU competitiveness and employment.

Future Environmental Costs and Benefits

The impact assessment was directed to establishing the relative environmental benefits of; first the various treatment options compared to landfill; and second of the recovery options against mechanical recycling. The impact assessment was based on an estimate of the change (increase or decrease) in the volume of materials per ELV treated between 2006 and 2015 by different methods, including landfill. Different estimates occur depending on the assumed nature of different treatment paths.

The environmental impact of increased recycling and recovery and the reduction in waste sent to landfill depends on a range of factors that includes:

- The material composition of the avoided waste, especially the size of the plastics fraction and the resin composition of the plastics
- The nature of the treatment option, especially for recovery (cement kiln, blast furnace, syngas production and waste incineration have all been examined)
- The efficiency of the recycling in terms of the amount of virgin material displaced for a given mass of recyclates (the substitution rate)
- The assumed levels and types of resources (energy or others) substituted in the case of recovery options
- The choice of how to compare the range of different impacts (the impact assessment has examined a range of impact categories)

The assessment was based on available LCA analyses for the different materials and especially for different vehicle components manufactured by plastics. The assessment is constrained by the absence of available data on all plastic resins and only a limited coverage of other materials (no data for glass, textile and rubber recycling for instance). This makes it difficult to provide definitive conclusions. Figures 2.1 and 2.2 summarise the available information from LCA for recycling and recovery respectively.

Figure 2.1: Coverage of ELV Materials by LCA Studies Relevant for Assessments of Recycling

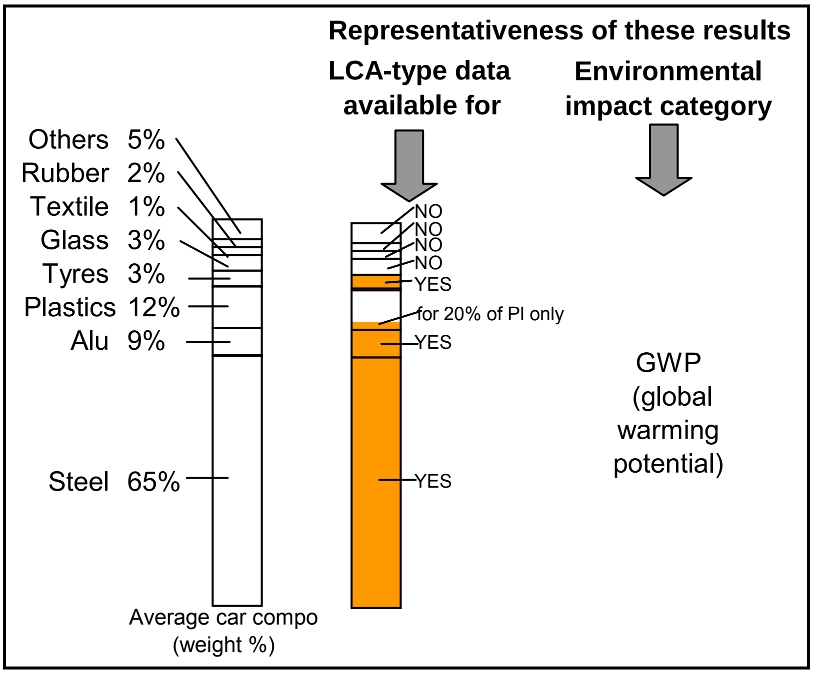
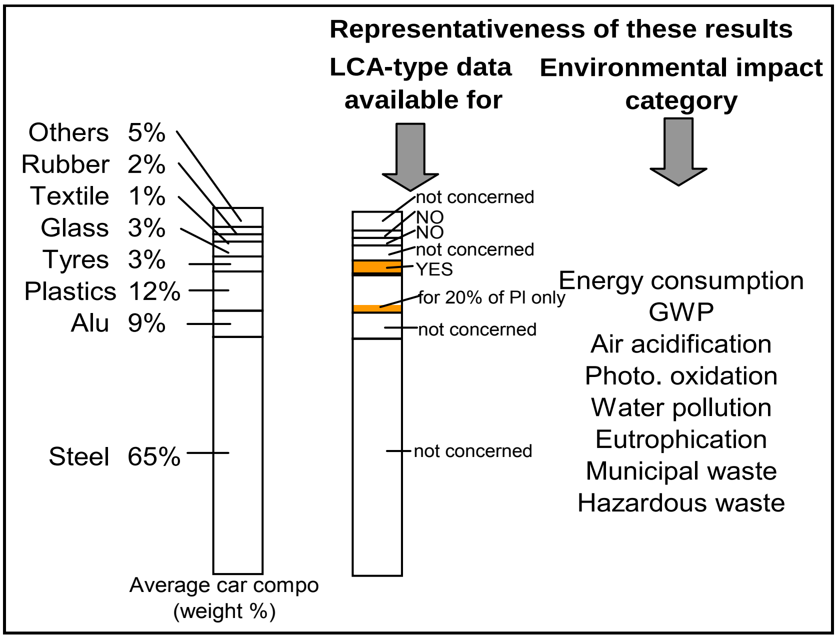


Figure 2.2: Coverage of ELV Materials by LCA Data Relevant for Assessments of Recovery



CONCLUSIONS

Costs per kg of ELV treated

The effects of changes in targets can be approximated by estimating the treatment cost per kg (for given compositions of material subject to treatment) of material diverted from landfill in 2015. Allowing for the changes in vehicle weight, composition and the assumed increase in real landfill costs and decrease in PST costs the marginal costs 2006-2015 (euro per kg of landfill avoided) range from 0.2 euro / kg (high cost scenario for thermal treatment) to -0.2 euro / kg (low cost scenario for mechanical separation) (Table 2.3).

Table 2.3: Marginal Costs of Landfill Avoided (euro / kg) – adjusting for vehicle weight, composition and assumed changes in unit costs

Options	Diversion (kg)	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
		Mechanical Separation		
Base – 2006	44	0.1	0.1	-0.1
Base – 2015	138	0.1	0.0	-0.2
2006 – 2015	94	0.0	0.0	-0.2
		Thermal Treatment		
Base – 2006	44	0.3	0.1	0.0
Base – 2015	138	0.2	0.1	0.0
2006 – 2015	94	0.2	0.0	-0.1

Note: Vehicle weight: Baseline: 951 kg. 2006: 964kg; 2015: 1025kg

Landfill volumes: Baseline: 180 kg; 2006: 145 kg; 2015: 51kg

These costs are driven by the assumed levels of unit costs (euro per kg ELV treated) for different treatment methods taking into account fixed and variable costs. Since these (with some adjustment) are the same to meet the 2006 targets as the 2015 targets the marginal costs are essentially constant. The unit costs do not change (within cost scenarios) with increasing levels of treatment; PST plants are assumed to operate at full capacity.

The effect of economies of scale is reflected in unit costs between cost scenarios, with higher unit costs associated with smaller plants in the high cost scenario and lower unit costs associated with larger plants in the low cost scenario.

The economies of scale and the high fixed costs associated with PSTs suggests that where these are employed to meet the 2006 targets, the additional costs of meeting the 2015 targets are much smaller than indicated by the assumed unit costs used in the cost calculation I Table 2.3, amounting to the variable costs of PST operation.

The costs per kg of landfill avoided (Table 2.3) assume that the capacity to meet higher targets beyond those for 2006 is brought on stream in line with the treatment of higher rates of recycling / recovery. In fact operators will (as illustrated in the technical review and summarised in Section 5) be able to secure overall recycling or recovery rates of at least 95%.Therefore it can be argued that the additional cost at plant level of meeting the higher targets beyond those for 2006 is equal to the net variable costs (gross variable costs of labour, energy, maintenance less sales revenue from recyclates or energy feedstocks).

These net variable costs vary between the PSTs but range from 22% to 53% of total costs per tonne, with three of the four technologies ranging from 42% to 53%. Assuming an average of 40% then the unit costs would be 40% lower than those applied to estimate the marginal costs above. The effect of excluding fixed costs is that the additional costs per ELV fall close to zero even in high cost scenarios because variable costs are offset by the savings in avoided landfill costs.

In other words if investment in PST is undertaken in order to meet the 2006 targets, then the cost of meeting higher targets is substantially less than those estimated, given the high fixed costs. This is likely to happen in several Member States. Based on the variable costs of PSTs, the marginal costs per kg fall to zero under the high cost scenario and produce savings under the other cost scenarios.

Net Costs of the Current 2015 ELV Targets

The additional costs per ELV of the higher targets in 2015 compared to 2006 are based on calculating the costs of meeting the targets in 2006 and in 2015 and subtracting the costs in 2006 from those in 2015. Three scenarios have been calculated, reflecting the range of costs identified. In the high cost scenario high treatment costs are combined with low landfill cost savings; in the low cost scenario low treatment costs are combined with high landfill cost savings and the medium cost scenario combines medium treatment cost estimates and landfill cost savings. In this way the high and low cost scenarios represent the full range of costs and cost savings.

The cost calculations indicate (Table 2.4) that the additional costs range from between 15 euro per ELV when thermal treatment is used, to a cost saving of 17 euro per ELV when mechanical separation is used. The medium cost estimate ranges from 5 euro to a saving of 4 euro per ELV. These results indicate that given the present information on the performance of treatment technologies, with mechanical treatment cheaper than thermal treatment, the higher recovery target of 95% would be met through a recycling rather than recovery approach. Given existing information on emerging mechanical recycling PSTs, they are effective up to 95%.

Table 2.4: Estimated Additional Cost of Achieving Higher Targets in 2015 (euro per ELV)

Options	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
Mechanical Separation	4.2	-3.5	-17.3
Mechanical & Thermal	14.6	4.6	-6.1

Impacts on Businesses

The additional costs per ELV identified imply an overall annual cost of up to 200 million euro (under the high cost scenario), with the possibility of cost savings of 240 million euro (under the low cost scenario).

Taking the high cost scenario based on variable costs, assuming capital costs have been incurred in order to meet the 2006 targets, there a limited cost impact of 40m euro on business of higher targets. In all other cost scenarios there are cost savings of up to 240 million euro per year.

The additional costs per ELV are small compared to the average new vehicle price. As a share of the price of a new vehicle the additional costs are very small (perhaps 0.2% in the case of the high cost scenario). The additional costs are also small (0.3%) when compared with the lifetime fuel costs of a vehicle.

The case studies indicate that at the present time ELVs prior to depollution have a positive value because of the scap metal. The present range of ELV scrap values are between 20 and 60 euro per ELV (excluding premature ELVs). However, the increase in the volume of metal in ELVs in 2015 is likely to add approximately 20 euro per ELV by 2015 at current scrap metal prices. To the extent that there is a positive value of an ELV after depollution costs (which may cost say 40 euro per ELV) some or all of any additional cost of treatment will be borne by the last vehicle owner. This suggests that even under the high cost scenario there is likely to be little or no charge on the free take back schemes.

There are unlikely to be any direct employment benefits or costs after taking into account the diversion of material from landfill operators.

Impacts on Competitiveness

There are only limited impacts on EU competitiveness and these are positive with some minor advantage to vehicle producers with a significant presence in the EU market from spreading any cost of take back schemes. There are also potential benefits for the treatment sector from the export of PSTs.

Environmental Impacts

The conclusions of the assessment of the environmental impacts of higher rates of recycling and recovery are considered in terms of the different treatment options for dealing with ELVs, comparing the relative impacts of higher recycling and recovery rates with landfill disposal, taking into account the impacts from the substitution of virgin materials.

1. *Recycling compared to landfill & recycling targets*

The assessment, taking Global warming Potential (GWP) as a representative impact category, indicates that:

- For ferrous and non ferrous metals recycling has environmental benefit from the substitution of virgin materials (with large benefits from avoiding producing hot rolled steel coil from uranium and aluminium ingot from bauxite). It also has benefits because of the avoided impacts from landfill.

- For plastics the recycling of some easily recyclable plastics can be beneficial compared to landfill (for some plastics a high substitution rate close to 1 is however a necessary condition – a specific study would be useful to further identify the resins concerned and the feasibility of recycling with such a high substitution rate). For other plastic pieces contained in ELV (and/or for low substitution rates), there is no evidence that recycling is beneficial compared to landfill. This is due to a combination of different effects: when landfilled plastics are not a major source of greenhouse gases and their recycling does not allow saving of enough virgin resources to compensate for the energy consumed during the reprocessing of plastics.

In terms of the environmental impacts from raising recycling levels above present levels and reducing the volumes disposed of to landfill, to reach higher treatment levels, plastics increase as a share of the treatment. Our analysis indicates that the increase in recycling of plastics and the associated possible disbenefits compared to landfill are outweighed by the benefits from the volume of metal treated.

As a consequence, when compared with landfill, recycling of ELVs is generally beneficial for GWP due to metal recycling, at least up to 95%. However, above a certain threshold (which is not possible to determine precisely but which is higher than 78%, i.e. the current market situation with no plastic recycled – and as a best guess would be around 85%), the level of net environmental benefits becomes less certain as the volume of plastics requiring treatment increases and the marginal benefits although positive, decline reflecting the treatment of more difficult resins, with lower substitution rates of virgin material.

2. Recovery compared to landfill & recovery targets

The impacts of recovery depend significantly on the treatment of plastics. However, the relative impacts of recovery against landfill are difficult to define. There are a range of relative positive and negative effects depending on the different recovery processes used and the nature of local circumstances in terms of the forms of energy and materials that the recovery process is deemed to substitute and also depending on the impact categories considered. Key parameters are also the type of energy recovered and substituted as well as the energy efficiency rate.

As a general rule, the research has identified (although with significant exceptions) that recovery provides greater benefits when it is undertaken using cement kilns, followed by blast furnaces (where use of materials is feedstock rather than energy recovery), followed by syngas production (also feedstock use). The results when recovering plastics using a MSW incineration option vary across impact categories: for most of the resins analysed, recovery with MSW incineration performs worse than landfill in GWP and hazardous wastes, but better in the other 6 categories assessed.

This result means that the environmental impact of higher rates of recovery of ELV fractions is uncertain because of the strong variability of the environmental profiles of plastic resins for the various recovery options.

If higher targets were associated with a requirement to employ certain techniques and applied only in the context of the substitution of certain forms of energy / materials generation, then there would be environmental benefits from higher rates of recovery. This is reflected in the recommendations.

These uncertainties also prevent any meaningful generalised conclusion about the benefits of recovery compared to recycling especially at higher rates where the relative benefits from metals recycling are less significant.

3. Summary conclusion

The increase in treatment to achieve higher rates of recycling or recovery is largely dependent on the increased treatment of plastics, for which relevant and peer approved LCA data is limited to less than 20% of the resins as used in vehicles. The importance of plastics and the absence of related data makes it difficult to formulate firm conclusions on the environmental benefits of increasing diversion from landfill. Additional work is needed on a wider range of resins as used in ELVs to confirm the indicative results presented above.

2.3 Recommendations

In this study, we were asked to analyse the benefits and costs of different possible targets for reuse, recycling, and recovery of ELVs in 2015. Having carried out this analysis, we recommend:

1. The target of 95% reuse, recycling and recovery is retained on the basis that there are environmental benefits to be gained at modest or zero cost.

2. Recycling would appear to be the more environmentally beneficial treatment, especially at lower levels of recycling because of the benefits from metal recycling. It also appears at the present time to be the cheapest option. It would therefore make sense to confirm the 95% target as a target for reuse and recycling, rather than recovery. However, at these higher rates the level and balance of environmental benefits between recycling and recovery options is less certain. Adoption of the target as a reuse and recycling target would have the effect of undermining the development of PSTs based on thermal treatment and the potential benefits that this technology might deliver. We therefore recommend in support of the target the establishment of a clear set of EU environmental criteria against which the PST options would be assessed when treating ASR allowing the selection of the most environmentally effective options. These criteria would have the benefit of:

    - Providing a clear and transparent basis for operators and regulators, with particular value to operators seeking in invest in more than one MS

    - Allowing scientific advance, including the results of current pilot technology testing and further LCA (especially of plastics), to inform future policy guidance

    - Removing the present uncertainty surrounding the distinction between recycling and recovery

    - (Possibly) allowing MS to fine tune the criteria to reflect national and local environmental circumstances to maximise environmental benefits, but applying the broad principles as reflected in EU guidance

    The downside of this approach is that is maintains some uncertainty and increased risk for technology developers, with the risk that insufficient capacity will be in place by 2015. However, with an investment lead time of say five years, there remains time to establish the criteria before major EU wide investment takes place.

3. The clear environmental benefits of recycling at lower rates justify the retention of the 85% recycling target, and because of the scope of most thermal treatment PSTs to undertake some mechanical recycling activity this target would not undermine their possible future development.

4. Partly as a counter balance to the retention of some continuing uncertainty over the specific environmental criteria, there should be a swift agreement to a 2015 target of 95% reuse, recycling and recovery to underpin investment, and to minimise the risk of a loss of sunk investment in landfill capacity.

5. A key commercial driver for PSTs is the availability of landfill and the level of landfill costs. To secure the most efficient response it is important that pressure is maintained to ensure that landfill costs fully reflect private and social costs – with the introduction of landfill bans on ASR when alternative capacity is available as a means of accelerating the introduction of PSTs.

6. The specific balance of costs and benefits depending on national and local circumstances means that MS should be given the flexibility to respond using a range of technical responses to take account of different costs and resources – this will be especially important for the smaller accession countries because of their lower labour costs and possible difficulties of achieving economies of scale from PST investment and hence higher treatment costs.

7. The Directive currently contains Articles which restrict the flexibility of the technical responses that might be developed, for example in relation to glass recycling. These Articles should be removed when the appropriate environmental criteria are approved.

PART B: CURRENT BENEFITS OF THE DIRECTIVE TO-DATE

3 ECONOMIC BENEFITS TO-DATE

3.1 Introduction

The introduction to the ELV Directive states that it aims firstly to minimise the impact of end-of life vehicles on the environment, thus contributing to the protection, preservation and improvement of the quality of the environment and energy conservation, and second, to ensure the smooth operation of the internal market and avoid distortions of competition in the Community. The primary aims of the Directive are therefore environmental, though the Directive aims to ensure that meeting these environmental aims is achieved in a consistent and non-distorting manner across the EU.

Any cost benefit analysis appraisal of the Directive needs to take account of the full range of economic benefits expected to result from its implementation. A core part of the benefit assessment relates to the environmental benefits of the Directive, which can be expected to account for a significant proportion of the overall benefits achieved, and which have an economic value. Some of these benefits can be valued at market prices (such as reductions in the costs of pollution damage or water treatment costs), while others may have no direct market value (such as reduced disamenity effects). While in theory the economic value of these environmental benefits can be measured through a combination of market prices, damage costs and non-market valuation techniques, in practice arriving at such valuations is problematic. The environmental benefits of the Directive to date are considered in Section 4, while the future environmental impacts and benefits of the Directive are addressed in Section 8.

As well as the economic values of its environmental impacts, the Directive can also be expected to give rise to a variety of further financial and economic benefits, through:

- Promotion of resource efficiency – potentially reducing raw material and energy costs by promoting recovery and reuse of valuable materials and components
- Promoting cost savings to public authorities, by reducing the costs of dealing with abandoned vehicles (through free take-back) and addressing the problem of vehicle crime and fraud (through an enhanced system of deregistration)
- Enhancing the efficiency and sustainability of the treatment sector, by raising professional and environmental standards and promoting modernisation of operations
- Reducing the costs of landfill to firms and the public, by increasing rates of reuse, recycling and recovery

These potential benefits are discussed in turn, based on the research undertaken, largely through the selected MS case studies.

3.2 Resource Efficiency

The reuse and recovery of valuable vehicle parts and materials is a long established practice, as old as the production of vehicles themselves. Many parts and materials, particularly metal ones, have sufficient value to make it cost effective to remove or recover them for reuse or recycling, and established markets and operations exist to achieve this. The Directive aims to increase rates of recycling, reuse and recovery further, both by introducing targets and by aiming to promote practices that facilitate dismantling and recovery.

Increased rates of reuse, recycling and recovery can yield environmental benefits by reducing disposal to landfill, and reducing the use of virgin materials and energy. They may also yield economic benefits where this results in a net overall cost saving to the economy.

However, given that the more valuable materials and components in ELVs were already recovered prior to the introduction of the Directive, we might initially expect the economic benefits of increasing rates of recovery to be small. Indeed, if we believed in perfect markets, imposition of mandatory targets which go beyond current practice would be expected to impose additional costs on the economy, by requiring recovery of parts and materials that might be more cheaply landfilled. So, any direct economic gains relating to resource efficiency need to come from the removal of market failure, for example, on the grounds that practices in operation prior to the Directive involved wasteful practices or sub-optimal levels of resource use. This might be the case if the systems that developed in the market place raised barriers to the recovery of parts and materials, even though higher rates of recovery might result in cost savings to the economy as a whole. A strong argument for this is that, prior to the Directive, vehicle manufacturers did not have the incentive to design and produce vehicles in such a way that potentially valuable components and materials could be easily recovered. While many parts and materials are relatively easily removed either by the dismantling or shredding process, it could be argued that some practices in vehicle manufacture (e.g. bonding of glass; mixing different plastics) obstruct the recovery of potentially valuable materials. Remedying market failure – in this case through changes in vehicle design and manufacture could bring about potential net efficiency gains.

A similar rationale applies in relation to the technologies available for treating Auto Shredder Reside (ASR - the material left after shredding and removal of most of the metal), which until now has been disposed of largely to landfill, but which contains potentially recoverable resources. With landfill costs relatively low, there has been little or no incentive to introduce alternative treatment technologies. Investments in technology have also been hindered by uncertainty about the demand for those technologies and the supply of materials for processing: By setting targets, the Directive has reduced the market failure due to uncertainty, leading to increased investment in technologies. The development of alternative treatment technologies (depending on costs and the value of recovered resources) reduces overall treatment costs.

The Directive contains a number of measures that potentially enhance resource efficiency. The introduction of producer responsibility ensures that manufacturers are now responsible for meeting the cost of take back and treatment of vehicles, and are therefore encouraged to have regard for reuse and recycling in the design and manufacture of vehicles, particularly when required to meet increasingly higher targets for recycling and recovery. Other provisions in the Directive also help to promote the recovery of materials and therefore potentially resource efficiency. These include the introduction of common standards for dismantlability, recoverability and recyclability in Article 7 (4), and the requirements of Article 8 with regard to coding of materials and components, and provision of dismantling information.

Manufacturers have responded to these requirements by developing vehicles that are more easily reused and recycled, by coding components and materials, and by developing dismantling information systems. For example, the International Dismantling Information System (IDIS) brings together 26 vehicle manufacturers who have developed a PC based information system to enable identification of component materials to promote more efficient treatment of end of life vehicles worldwide. Individual manufacturers have also made progress in designing vehicles to promote recycling. For example, Renault won a ecodesign award for the Modus dashboard on account of its high degree of recyclability as well as its recycled content[2], while Toyota has introduced a "Design for Recycling" approach, developed new recyclable materials, and introduced an "Easy to Dismantle" mark[3].

With the costs of take-back and processing depending on the efficiency of the processing facilities, manufacturers also have an incentive to increase the efficiency of processing facilities. Manufacturers have set up networks of authorised treatment facilities across the EU – and there is evidence that, to reduce costs, they have promoted the most efficient practices in those facilities. This is another method by which the Directive reduces pre-existing market inefficiencies.

Because of the time difference between the manufacture of vehicles and their treatment as ELVs, as well as the time taken to redesign vehicles, the economic benefits resulting from design are likely to be realised after 2010. It has been reported that the take back provisions have led vehicle producers to revise design such that the non-ferrous metal content has been increased (for example at the expense of certain plastic components in door panels) to increase the value of ELVs and hence to finance the costs involved in operating the take back scheme.

3.3 Economic Benefits to Public Authorities

The Directive has brought benefits to public authorities in the EU by helping to deal with the problem of abandoned vehicles and to tackle vehicle crime. The required introduction of free take back of end of life vehicles should help to reduce the problems of abandonment and illegal disposal, which typically impose costs on local authorities who are then required to dispose of the vehicles concerned. Though abandoned vehicles are a problem across the EU, the only evidence available documenting these benefits is from the UK.

The requirement to issue Certificates of Destruction as a condition for deregistration should improve information about the vehicle stock in those countries where such a system did not previously exist. It enables the vehicle licensing authorities to establish accurate records of vehicles that have reached the end of their lives and been disposed of. The need for a licensing system for treatment facilities may also help the authorities to tackle vehicle crime.

Dealing with abandoned vehicles imposes significant costs on local authorities. For example, in the UK, the Chartered Institute of Wastes Management (CIWM) estimated that 221,000 vehicles were abandoned in 2002/03, typically resulting in a cost to the local authority of £360 (525 euro) per vehicle to arrange environmentally friendly disposal (CIWM, undated). Rates of vehicle abandonment increased substantially in the early years of the current decade because of a collapse in scrap metal prices, giving ELVs zero or negative value.

A report by TRL (2003) found that the costs of dealing with abandoned vehicles varied widely between vehicles and between different parts of the UK, but were increasing due to low scrap values at the time. Average charges by contractors to collect abandoned ELVs varied between 0 and £150 (220 euro) per vehicle in the UK, averaging £30 (45 euro) per vehicle, with the total costs of dealing with an ELV (including storage, officer time and disposal) put at between 25 euro and 370 euro per vehicle.

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[2] http://www.dexigner.com/forum/index.php?showtopic=3741

[3] http://www.toyota-europe.com/images/Chapter_7.pdf

While the Directive is expected to reduce the abandoned vehicle problem following the requirement of free take back in 2007, there has been some concern that higher treatment standards, which have raised the cost of dealing with ELVs, may have increased the problem in the interim period. For example, in the UK, estimates by the Regulatory Impact Assessment and the House of Commons Trade and Industry Select Committee report on the Directive suggested that the number of abandoned vehicles could increase between 150,000 and 500,000 before free take back. However, the real increase in scrap metal prices since the assessment has reduced this possibility.

With regard to vehicle crime, the only available evidence again relates to the UK where the Government has predicted that the requirement to issue CoDs will reduce the risks of car crime and fraud by improving the effectiveness of the vehicle registration system, as part of a new electronic system for continuous registration of vehicles. The Driver Vehicle Licensing Agency (DVLA) estimated in August 2004 that an additional 400,000 vehicles had already been taxed under the new system.

In particular, CoDs will make it more difficult to use scrapped vehicles to disguise the identity of stolen vehicles, which has been a common problem in the past. More accurate vehicle registers help to reduce the scope for running vehicles illegally, for using unregistered vehicles to undertake crime, and for abandoning vehicles with impunity. CoDs are part of a wider package of measures to tackle car crime that the UK Home Office has estimated could yield savings valued at up to £200 million (280 million euro) per annum. A report by the National Audit Office[4] presented evidence to the UK's Regulatory Impact Assessment on the ELV Directive suggesting that changes in vehicle registration and checking procedures had contributed to a 13% reduction in the number of stolen vehicles between 2002/03 and 2003/04. Although systematic data on car crime and the scope to achieve the UK effects in the EU25 has not been collected it suggests significant EU benefits are possible.

3.4 Efficiency and Sustainability of the Treatment Sector

The Directive has required significant investment in collection, storage and treatment facilities for ELVs. The significant costs involved, and the requirement for all facilities storing and treating undepolluted ELVs to obtain a permit, have led to a significant rationalisation of the treatment sector as well as a significant increase in standards. As a result, the case studies suggest that the vehicle treatment sector is now widely regarded as being more efficient, professional and sustainable as a result of the Directive. The Directive has also provided a more certain environment in which the treatment sector can plan the development of capacity, which will be determined by recycling targets and less sensitive to market factors such as fluctuations in commodity prices. As a result of this combination of factors, the Directive can be expected to promote economies of scale in treatment and recycling, and to encourage investment and innovation within the sector.

The costs of implementing the Directive therefore need to be viewed against the benefits of investment in raising standards, enhancing the image of the sector, improving efficiency and innovation, and securing its longer term future. These impacts are considered in more detail in Part C in the context of longer term costs and benefits.

3.5 Reductions in Landfill Costs

By diverting ELV waste away from landfill and towards reuse, recycling and recovery, the Directive has reduced landfill costs across the EU. These cost savings need to be

[4] http://www.nao.org.uk/publications/nao_reports/04-05/0405183es.pdf

compared to the additional costs of dismantling, reprocessing and recovery of ELVs, in assessing the net costs and benefits of the Directive. On the basis that approximately 25% of the weight of ELVs was landfilled prior to the Directive, meeting the 85% target for reuse, recycling and recovery in 2006 will reduce the cost of landfill of ELVs by approximately 40%, while meeting the 95% target by 2015 will reduce landfill costs by 80%.

On the basis that around 10 million tonnes of ELV waste are currently produced in the EU with 25% landfilled, and using an average landfill cost of 80 euro per tonne, meeting the 2006 target reduces landfill costs by 80 million euro per year, with further annual savings of 80 million euro per year from meeting the 2015 target.

With the capacity of landfill limited in many Member States, filling landfill with ELV waste - which can be recycled or recovered - reduces future landfill capacity, increasing future landfill costs for waste that would be more expensive to divert to alternative treatment or disposal routes. Diversion of ELV waste to other treatment routes therefore brings additional, 'external' cost reductions for other future waste streams.

The cost assessment in Part C takes account of the savings in landfill costs in assessing the net costs of implementing the Directive.

4 ENVIRONMENTAL BENEFITS TO-DATE

4.1 Introduction

Potential ELV environmental impacts fall into two main categories: pollution and material loss.

Possible sources of environmental impacts within these categories are: landfilling of waste from shredders, poor environmental practices at some auto dismantlers and vehicles abandoned in the environment.

Materials with potential negative environmental consequences in ELVs include: oil, coolant, fuels, brakes and other fluids; heavy metals including lead (Ld), mercury (Hg), cadmium (Cd), chromium VI (Cr(VI)). The second category of potential environmental impacts relate to waste and resource loss from not maximising ELV reuse, material recycling and recovery.

A key prerequisite for improving the environment outcomes is better management of the process by which ELVs enter the waste stream (e.g., through deregistration requirement) in particular to reduce abandonment or ELV being sent to informal facilities. Environmentally sound management of ELV disposal is then the next step to reduce environmental impacts.

The ELV directive wanted to address these issues.

4.2 Objective of this section

The question we want to answer in this chapter is: **What are the main environmental benefits of the directive to-date?**

- ‘To-date’ i.e. considering the current practice on the ground (which are not necessarily in full compliance with the Directive yet)
- ‘Of the directive’ i.e. focusing on the changes in behaviour that can be attributed to the implementation of the ELV directive.

4.3 Scope of the analysis

In the ELV Directive, the minimum technical requirements for treatment include:

1. Sites for storage (including temporary storage) of end-of-life vehicles prior to their treatment:
   - Impermeable surfaces for appropriate areas with the provision of spillage collection facilities, decanters and cleanser-degreasers
   - Equipment for the treatment of water, including rainwater, in compliance with health and environmental regulations

2. Sites for treatment:
   - Impermeable surfaces for appropriate areas with the provision of spillage collection facilities, decanters and cleanser-degreasers
   - Appropriate storage for dismantled spare parts, including impermeable storage for oil-contaminated spare parts, appropriate containers for storage of batteries

(with electrolyte neutralisation on site or elsewhere), filters and PCB/PCT-containing condensers

- Appropriate storage tanks for the segregated storage of end-of-life vehicle fluids: fuel, motor oil, gearbox oil, transmission oil, hydraulic oil, cooling liquids, antifreeze, brake fluids, battery acids, air-conditioning system fluids and any other fluid contained in the end-of-life vehicle
- Equipment for the treatment of water, including rainwater, in compliance with health and environmental regulations
- Appropriate storage for used tyres, including the prevention of fire hazards and excessive stockpiling

3. Treatment operations for depollution of end-of-life vehicles:
   - Removal of batteries and liquified gas tanks
   - Removal or neutralisation of potential explosive components, (e.g., air bags)
   - Removal and separate collection and storage of fuel, motor oil, transmission oil, gearbox oil, hydraulic oil, cooling liquids, antifreeze, brake fluids, air-conditioning system fluids and any other fluid contained in the end-of-life vehicle, unless they are necessary for the re-use of the parts concerned
   - Removal, as far as feasible, of all components identified as containing mercury
4. Treatment operations in order to promote recycling:
   - Removal of catalysts
   - Removal of metal components containing copper, aluminium and magnesium if these metals are not segregated in the shredding process
   - Removal of tyres and large plastic components (bumpers, dashboard, fluid containers, etc), if these materials are not segregated in the shredding process in such a way that they can be effectively recycled as materials
   - Removal of glass

In this section we successively analyse the 9 different fractions concerned by these ELV management requirements, first those concerned by depollution issues then the others:

- Fluids
- Batteries
- Liquefied gas tank
- Air bags
- Catalysts
- Glass
- Tyres
- Large plastic components
- Metal components

Another requirement concerns the collection and transfer of ELV to authorised treatment facilities (ATF). It constitutes the 1st topic analysed in this section.

Remark: another requirement of the directive concerns the phasing out of certain heavy metals from cars put on the market after 1 July 2003. After preliminary investigation showing that no literature or statistics are available to explore this large issue, it was decided to leave it out of the analysis. Important efforts would be necessary to gather useful information, which can not be done in the framework of this limited study. Whatever, the potential environmental benefits occurring at the end-of-life stage are likely not to be visible yet as most of the vehicles put on the market in 2003 will be treated within 10-15 years.

4.4 Methodology developed

A fraction-based approach

Because practices and potential environmental impacts and benefits differ according to the fraction, the assessment was performed for each fraction separately.

A 2-step analysis: full compliance then current practice

As current practices are not necessarily in full compliance with the ELV directive yet, we performed first the assessment of environmental benefits in the case of full compliance then we discussed the results to assess those to-date.

The assessment of the benefits to-date was not an easy exercise given that (i) current practices are not harmonised across Europe, (ii) there is no reporting from MSs to the EC and (iii) the literature is not prolix on this issue.

Apart from the illegal dismantlers which are still numerous, there are ATF which have already an authorisation in compliance with the ELV directive and the others which are still in the process of being delivered an additional authorisation. Practices in Europe are not homogeneous between these categories and even not necessarily within each category as it is further analysed in section 0 below.

Baseline scenario

In order to focus on the changes in behaviour that can be attributed to the implementation of the ELV directive, a baseline scenario was defined corresponding to the practices which would occur if the directive would not exist.

To build the baseline scenario, we **started from the practices existing prior to the directive and made assumption on how they would have evolved without the ELV Directive** but considering the other possible drivers (other legislation pieces, existing evolution trends, voluntary agreements…).

A different baseline scenario had to be defined for each fraction.

This exercise was not easy either, in particular because of the interconnection of different existing directives (Waste Oil, Batteries, Landfill directives…) which are themselves more or less well implemented.

Two sources of environmental benefits and impacts

In order to provide a comprehensive response to the question considered, we did not only consider the **direct benefits of the avoided pollution** corresponding to the previous practice (e.g., soil contamination by fluids spillage) but we also tried to integrate the **environmental benefits (and impacts) of the new practice** (e.g., those due to the regeneration of or energy recovery of fluids).

We then considered two types of environmental impacts or benefits:

- **Local** environmental impacts or benefits, e.g. those occurring at the dismantling or shredding facilities
- **Global** environmental impacts or benefits, i.e. those occurring at different stages of the process and at different periods in time

Two main tools are available to assess these environmental impacts:

- **Environmental Risk Assessments (ERA):** the purpose of a risk assessment is to determine the risk posed by a chemical or a chemical product to human health and to the environment. Key parameters are dose-response couple and exposure. A brief presentation is included next page.
- **Life Cycle Assessments (LCA):** this is a tool for the systematic evaluation of the environmental aspects of a product or service system through all stages of its life cycle. LCA captures global impacts. A brief description of key steps is attached in section 8.1.3.

Sources of information

Different sources of information were used to perform this assessment:

- Changes of behaviours due to the directive: in the absence of relevant reporting from the MSs and comprehensive literature on this issue, important efforts were put on trying to get an idea of:
  - current practices at the European level
  - previous practices at the European level
  - changes which can be attributed to the ELV Directive

  Interviews with key stakeholders (enforcement officials and dismantlers / shredders industry representatives either from federations at the European or national levels or from individual companies). At the national level, we focused on the countries selected for the case studies: D, F, HU, NL, PL, UK.
- Local environmental impacts and benefits: sources of information include these same interviews, literature review (incl. risk assessments available) and BIO's own expertise.
- Global environmental impacts and benefits: LCA and other environmental data available in the literature as well as BIO's own expertise and in-house database.

Caveats: in this study, **it was of course not planned to carry out any ERA or LCA. Instead we performed an extensive literature review to identify and select existing ones (and preferably peer-reviewed international ones)** from which we extracted key results in order to document our analysis. Used references are given hereafter for each concerned topic.

A similar structure for the analysis of the 11 topics

Each of the 11 following sections follows a similar structure:

a/ Reminder of the requirements of the Directive

b/ Current practice

c/ Main practices prior to the Directive and main environmental problems associated

d/ What would be the practice if no ELV Directive (baseline)

e/ Potential benefits in case of full compliance with the ELV Directive

f/ Benefits to-date due to the ELV Directive

Box 1: Brief presentation of Environmental Risk Assessment methodology

For chemical substances, the scope of Environmental Risk Assessment (ERA) covers emissions and consequent environmental impact and human exposures at each stage of the life-cycle of a chemical, including production, processing, formulation and use, recycling and disposal. Besides human health, protection goals for the environment include the atmosphere, aquatic organisms, sediment dwelling organisms, soil-dwelling organisms, micro-organisms in waste water treatment plants, and mammals and birds exposed via accumulation up the food chain.

For each individual chemical, the assessment of its potential risks consists of three main steps:

1. effect assessment (hazard identification; dose-response assessment)
2. exposure assessment
3. risk characterisation

1) Effect assessment

- *Human health effects assessments* should be conducted for a number of effects: Acute toxicity, Irritation, Corrosivity, Sensitisation, Repeated dose toxicity, Mutagenicity, Carcinogenicity and Toxicity to reproduction.

For each of these endpoints, a NOAEL (No Observed Adverse Effect Level) should be determined. If a NOAEL can not be determined, then a LOAEL (Lowest Observed Adverse Effect Level) should be identified.

- *Environmental effects assessment:* a PNEC (Predicted No Effect Concentration) should be determined for a broad range of species representative for the environmental compartment under investigation. The calculations are based on a variety ecotoxicity data (acute and chronic), environmental fate and physical properties of the chemicals.

2) Exposure assessment

- Human Exposure

Direct exposure measurements and environmental modelling are used to assess the levels to which humans are exposed to a chemical via air, drinking water and food in the environment, through consumers' products and in working places. This leads to a large number of exposure scenarios, for which risks must be individually assessed.

There is often a lack of information regarding all these exposure sources and hence it is often difficult to carry out a reliable exposure assessment.

- Environmental Concentrations (PEC)

Using basic data on the volume of a substance produced or processed and the estimated releases to the environment, properties of the chemical (e.g., volatility, water solubility, (bio)degradation, and partitioning behaviour between water and air), the environmental distribution of a chemical can be modelled.

This results in a series of Predicted Environmental Concentrations (PEC) at each industrial site, and also over a defined region, for each environmental compartment (air, water, soil).

3) Risk characterisation

- Human populations

Risk = No Observed Adverse Effects Level (NOAEL) / Exposure

The ratio gives a Margin of Safety (MoS). A judgement is then required on the sufficiency of the MoS. Either the margin is sufficiently large to conclude there is no concern, or so narrow that a risk to health cannot be excluded.

- Environment

Risk = Predicted Environmental Concentration / Predicted No Effect Concentration (PNEC)

Where the predicted no effect concentration is exceeded, i.e. PEC/PNEC>1, it is considered that there is a risk.

With regard to **global ecosystem risk assessment**, it is important to bear in mind that point source releases may have a major impact on the environmental concentration on a local scale and contribute to the environmental concentrations on a larger scale. Thus the chemical in question may have important local impacts but also regional and global impacts. This is particularly true for those substances whose presence in the environment directly or indirectly impacts the major biogeochemical cycles (e.g., oxygen, carbon, nitrogen, phosphorous cycles) and thus enhances the global environmental changes such as climate change.

4.5 Topic 1 - Collection and treatment in authorised treatment facilities

Key figure	10.8 millions ELVs[5] requiring treatment in EU25 in 2005

a/ Requirements of the Directive

Articles 5.1 and 5.2 of the Directive require that 100% of ELV are collected and transferred to authorised treatment facilities (ATF).

b/ Current situation in MSs

This objective is clearly not reached in most Member States yet:

- The number of illegal dismantling facilities (with no environmental control) is still relatively high.

  Two types of facilities exist in Europe:

  - authorised treatment facilities (ATF): as mention in section 5.1, there are about 8,000 authorised dismantlers and 232 shredders in Eu25.

    Figures related to authorised dismantlers are orders of magnitude as many countries do not have an efficient monitoring system in place yet (for instance in F and UK, the total number of ATF is not known, authorisations being delivered by local authorities with no systematic reporting made to national authorities).

    Remark: this lack of efficient reporting / monitoring system is also linked to the fact that the permitting process specific to the ELV directive in not fully operational yet (and actually in F and the UK for instance, the term 'ATF' refer to installations authorised to operate prior to the ELV directive when reporting / monitoring system were not considered crucial) – See below for more details.

  - Illegal facilities: there are no statistics available either at the European level or at the national level to assess the number of illegal facilities precisely. But stakeholders converge to say that there are important discrepancies between countries.

    According to EGARA and ARN (NL), in MSs where there is a funded system (such as NL, Dk, Sw) for quite a long time (e.g., more than 10 years in the NL), illegal dismantlers have progressively being replaced by ATF (in particular in order to be able to beneficiate from financial support). The number of illegal facilities is believed to be lower than 10% in such countries (or even close to 0%).

    Based on estimations provided by CNPA (French dismantling organisation), the number of illegal facilities would still reach 40% in France (800-900 illegal vs. 1,200-1,300 ATF). In Hungary, it would be around 80% (Car-rec indicated several hundreds of illegal facilities vs. 80 ATF)[6].

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5 GHK estimate – see table 7.1 section 7.1

6 According to a treatment operator located in Belgium, there would be around 50% of illegal facilities in Belgium. Commercial practices of importers over the last 6 years explain (at least partly) the decrease of the number of informal dismantlers during that period. In the absence of car makers in Belgium, importers are the ones who compete to gain market share amongst car dealers. To do that, they offer car dealers, for each new car sold, to pay for the cost to send an old car to

- Thus the number of ELV not treated in ATF is still relatively high.

  From data provided in a previous section of the report, 4-6 millions[7] of ELVs can be estimated being treated in ATF in 2005. Considering that about 10.8 millions[8] ELVs require treatment in EU25, then **about 50% of ELVs would be treated in ATF**.

  Remark: stakeholders interviewed in selected MSs converge in saying that remaining illegal facilities have usually smaller capacities than authorised ones. This can be explained by different factors including: control is easier with bigger facilities; bigger facilities, often belonging to large waste management groups, are more easily able to comply with the legislation. As a consequence, the proportion of ELV being transferred to ATF is higher than the proportion of ATF. Thus more than 50% of dismantling facilities would still be illegal in Europe in average.

Remark 1 concerning ATF:
'ATF' is an ambiguous term which revealed, during our research, not to designate the same type of authorisation according to stakeholders and MSs. Actually, it seems appropriate to distinguish between 2 types of ATF:

- ATF which comply with the ELV directive
- ATF which do not fully comply with the directive yet

Focusing on our case studies will help illustrating this point.

Table 4.1: Dismantling facilities in MSs under study

	No. of ATF		Total number of ELV requiring treatment / yr	Number of ELV treated in ATF [9]	Number of illegal facilities [10]
	With a permit **in line** with the ELV directive	With a permit **not yet in line** with the ELV directive			
D	1178 [11] (See Box A)		?	1,200,000	Few?
F [12]	0	1200-1300 (out of which 450 certified) (See Box B)	?	1,300,000	800-900
HU [13]	80	?	100-120,000	1,500	Several hundreds
NL [14]	475 (See Box C)	0	272,000	272,000	0

accredited centres. This practice encourages car dealers to take back old vehicles (even if not yet ELV) and to send them to ATF.

7 ∑ (No. of ATF per MS x No. treated ELVs per ATF); Source for 'No. of ATF per MS' and 'No. treated ELVs per ATF': ACEA, 2005 (2004 data) - See table 7.3 section 7.4.2

8 GHK estimate – see table 7.1 section 7.1

9 Source: ACEA completed by EGARA, Dec 2005 (2004 data)

10 No authorisation to operate at all (even prior to the ELV directive)

11 Source: UBA, Dec 2005; ACEA, 2005

12 Source: CNPA and French Environment Agency Ademe, Dec 2005

13 Car-rec, Sept 2005

14 Source: ARN, Dec 2005

PL	Few?	670 [15] (See Box D)	?	80,000	~1,500 [16]
UK	0?	~2,500[17] out of which 1,088 licensed (See Box E) [18]	?	2,110,000	500-800 [19]

Box 2: Germany

The permitting process followed the 1998 ELV Ordinance in Germany (prior to the 2002 ELV Act transposing the ELV directive).

Box 3: France

No statistics are available in France about ATF.

'ATF' refer to facilities with a permit from the competent authority in compliance with Art 5 of the Waste Directive 75/442/EEC; an additional authorisation will be delivered to these ATF by authorities when they fully comply with the ELV directive; it is believed that the majority of these additional authorisations will be delivered by the end of 2006.

But amongst these 1,200-1,300 ATF, about 450 are certified as of Dec 2005. As soon as 1993, the French dismantling organisation (CNPA), anticipating the ELV directive, developed a certification process which is operational since 1995. This certification[20] consists in a tailor-made environmental management system which is certified by a third party. Certification criteria include the ELV directive requirements about depollution. The removal of pieces for recovery (glass, tyres, plastics pieces…) is encouraged if economically feasible.

Box 4: Netherlands

The 475 ATF existing in the NL received an environmental license from the government authorizing them to operate in accordance with the ELV directive. This permitting process began as early as 1995 with a national legislation anticipating the ELV directive.

Remark: 275 ATF are affiliated to ARN, receiving about 90% of ELV treated in the NL; about 200 others are not affiliated to ARN, representing around 10% of ELV. To be affiliated to ARN, an ATF needs an additional third-party certification (dealing with working conditions, quality issues, etc.).

According to ARN, a combination of different drivers explain that no illegal facilities operate in the NL: (i) a very strict control body, (ii) an efficient deregistration system (incl. the existence of an ownership tax since 1997; the only way to stop paying this tax is to prove that you are no

15 Source: ACEA, 2005; figure compatible with those published in www.geft.org/file/toolmanager/050F24282.htm (500 authorised + ~200 applying for authorisation in 2001)

16 Source: www.geft.org/file/toolmanager/050F24282.htm (2001 data)

17 Source: DTI (2003) estimated that there are 2,500 dismantlers, salvage operators, scrap yards and secondary metal merchants currently dealing with ELVs, typically SMEs (see UK case study in Appendix)

18 Source: UK Environment Agency, Dec 2005 – this figure increases rapidly; the figure of 732 which can still be found in many reports corresponds to the situation a few months ago

19 Source: DTI, 2003 (see UK case study in Appendix)

20 Référentiel certification de services Qualicert n° RE/REM/03 entitled 'Traitement et valorisation des véhicules hors d'usage et de leurs composants'

more the official owner of the car, which can only be proven by a COD delivered by an ATF).

Box 5: Poland

Following the publication of the Polish ELV Act in April 2005, dismantlers have to obtain a new decision from local authorities giving permission to operate in accordance with the ELV directive; 1st permits begun to be delivered in Sept 2005 and this process will go on in 2006.

No statistics about the number of ATF complying with the directive will be available before probably end 2006 according to the National Fund for Environmental Protection and Water Management, which expects that about only 200 big dismantlers will remain (others not being able to absorb upgrading costs).

Box 6: UK

2 types of dismantlers exist in the UK:

- those registered,
- those registered and licensed, i.e. having received an additional 'waste management license' (a permit to use land); there are 1,088 licensed sites as of Dec 2005.

To be in compliance with the ELV directive:

- registered installations must first received a waste management license,
- licensed installations have to make the necessary investment to upgrade their equipment and procedures (e.g. parking spaces can be hard-packed surface not impermeabilised yet; water treatment may need to have a cleanser-degreaser installed; etc.) in order to have the terms of their license modified accordingly.

Our understanding is that since Nov 2003 (ELV directive implementation in the UK), there is an increasing number of registered dismantlers obtaining a license, which is not linked to the ELV directive itself and which should have occurred before. The modification of their license (when in compliance with the ELV directive) is the next step, which began recently and will pursue in 2006. No data are available regarding the number of licensed dismantlers already in compliance with the ELV directive. It seems reasonable to assume that the majority of them are not necessarily in full compliance with the ELV directive yet.

Let's summarise key lessons from the table and boxes here above.

In D and the NL, national legislation anticipated the ELV directive in 1997-1998 and implemented a permitting procedure for dismantling facilities. 'ATF' thus designates, in these countries, installations complying with the ELV directive.

In F, HU, Pl, S and the UK, the implementation of the ELV directive began only recently. **'ATF' refers to installations with a waste management permit existing prior to the ELV directive**. **The delivery of additional permits (or the revision of existing permits according to the country) is going on** (or will start soon). The upgrading of dismantling facilities to ELV requirements will require more or less investment according to the current situation / practice of these installations. For instance in France, CNPA and ADEME assume that certified ATF will require less investments to comply with the directive compared to most of non certified ones as efforts have already been made on a voluntary basis.

Remark 2 concerning ATF:

Another issue worthwhile to be mentioned concerns the type of requirements ATF have to comply with (in the case of permits in line with the ELV directive).

According to stakeholders interviewed in MSs and at the EU level, in lots of (all?) cases, these requirements include all the depollution requirements of the directive, but not always all the recycling requirements (in particular glass and large plastic components removal for recycling purposes; e.g. Germany recently abandoned the obligation to remove glass for recycling because of costs involved).

In the absence of adequate reporting about this issue, a validation of this information can not be done in the framework of this study with limited resources.

c/ Situation prior to ELV directive and main environmental problems associated

Although precise figures are not available, it has been estimated that vehicles abandoned in the environment may account for up to 7% of total ELVs in certain MSs[21].

Besides, the ELV treatment sector is well-known for having been dominated by uncontrolled practices. Poor practices in illegal facilities and the environmental problem associated are described for each fraction in the next sections.

In facilities authorised according to national waste legislation, as well as in illegal ones, **practices have been market-driven: only what had a value was removed and recovered**. Traditionally, ELV have undergone relatively high levels of recycling (they are one of the most highly recycled consumer goods). Components having an economic value were removed by dismantlers or after shredding for refurbishing, reuse, recycling or energy recovery (in particular ELV contain up to 75% ferrous metal, which is easily recycled). The remainder of the ELV, primarily glass, plastics, seat foam and rubber, was send to landfill as waste.

d/ What if no ELV Directive

Without the Directive, there would have been no major incentive to modify the type of treatment facilities and practices. There would still be a majority of illegal treatment installations as today in MSs which are late in implementing European legislation or where regulation enforcement is lacking or poor. And practices would be still mostly market-driven.

However, in some MSs where environmental consciousness is more developed, it is likely that environmental management systems would have developed amongst dismantlers and shredders with ISO 14001 or EMAS being made available and promoted (without being able to estimate to which extent; however, according to some stakeholders, it would have concerned a minority of them in most of the MSs[22]).

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21 Commission proposal COM (97) 358 Final, July 1997 (page 3)

22 An upper bound could be the current number of dismantlers and shredders awarded EMAS, which is lower than 177 (i.e. less than 2% of a total of 8232 dismantlers & shredders hold EMAS). Explanation: in November 2005, out of the 3175 EMAS holders, 177 of them were with a 90 code NACE i.e. 'Sewage and refuse disposal, sanitation and similar activities'; this category is larger than

e/ Potential benefits in case of full compliance with the ELV Directive

From a theoretical point of view, 100% ELV would be treated in ATF, with no more abandoned vehicles in the environment and no more poor practices.

And market-driven practices would no more dominate: materials with no or negative value would also be properly treated or recycled / recovered. More specific benefits are assessed for each fraction (see next sections).

f/ Benefits to-date due to the ELV Directive

Three main benefits can be identified at that stage:

There is a slow but real progressive replacement of illegal treatment facilities by authorised (and thus controlled) ones. Time constant is long but the trend is there. Some interviewees estimate that it may take another 10 years to have all illegal treatments disappear thanks to the Directive (part of the difficulty is that these small installations play locally an economical role; most of them are not run by criminals; with the directive, it is believed that local authorities will be more and more entitled to enforce the law even if this generates local economical difficulties).

Even if authorised dismantlers do not all comply with the directive yet (as part of the ELV directive implementation process, the permit delivery is still in progress in most MSs), the number of ATF adopting the environmental requirements of the Directive has been increasing regularly (since even before 2002 by anticipation of the Directive in several MSs, either through voluntary agreements between dismantlers, shredders and car makers and third party environmental certification or through a national legislation).

Proper treatment and recycling / recovery of not only materials with a positive value but also more and more of those with a 0 or negative economic market value, including the development of new markets (this is further discussed for each fraction hereafter).

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dismantlers and shredders since it covers other waste treatment activities as well as water treatment activities.

4.6 Topic 2 - Depollution of ELV: fluids

Key figures

	(1)	Total in ELV requiring treatment	Total in ELV treated in ATF
Engine oil	*2,86 kg/ELV*		
Transmission oil	*2,06 kg/ELV*		
Suspension oil	*0,58 kg/ELV*		
Brake fluid	*0,37 kg/ELV*		
Oil filter oil	*0,14 kg/ELV*		
Power steering	*0,09 kg/ELV*	(2)	(3)
Sub-total **oil**	6,10 kg/ELV	**66 kt in Eu25/yr**	**31 kt in Eu25/yr**
Coolant	*3,43 kg/ELV*		
Screenwash	*1,60 kg/ELV*		
Sub-total **water-based fluids**	5,03 kg/ELV	**54 kt in Eu25/yr**	**25 kt in Eu25/yr**
Sub-total **fuel**	11,29 kg/ELV	**122 kt in Eu25/yr**	**56 kt in Eu25/yr**
Total **fluids**	22,42 kg/ELV	**242 kt in Eu25/yr**	**112 kt in Eu25/yr**

(1) Source: 'A Study to Determine the Metallic Fraction Recovered from ELV in the UK', Jema Associates Ltd & David Hulse Consultancy Ltd, for DTI, Sept 2005 (p12)

(2) 10.8 millions ELV to be treated in Eu25 in 2005

(3) 4-6 millions ELV treated in ATF in 2005; 5 millions considered for the calculations

a/ Requirements of the Directive

Annex 1 of the Directive requires the following:

Sites for storage (including temporary storage) of end-of-life vehicles prior to their treatment:

- Impermeable surfaces for appropriate areas with the provision of spillage collection facilities, decanters and cleanser-degreasers
- Equipment for the treatment of water, including rainwater, in compliance with health and environmental regulations

Sites for treatment:

- Appropriate storage tanks for the segregated storage of end-of-life vehicle fluids: fuel, motor oil, gearbox oil, transmission oil, hydraulic oil, cooling liquids, antifreeze, brake fluids, battery acids, air-conditioning system fluids and any other fluid contained in the end-of-life vehicle
- Impermeable surfaces for appropriate areas with the provision of spillage collection facilities, decanters and cleanser-degreasers
- Equipment for the treatment of water, including rainwater, in compliance with health and environmental regulations

b/ Current practices

Vehicle operating fluids, including fuels, represent about 2% of an average car composition (about 22 kg).

Fuels are generally separated from other fluids (even by illegal operators) as they have an economical value and can be easily reused on-site.

The remaining fluids (waste oils and water-based fluids) represent about 11 kg per ELV.

In AFT in line with the ELV directive, different practices exist differing by the number of flows separated and their destination. For instance:

- Practice A - separate storage of 5-6 fractions which are sent to different reprocessing or regeneration facilities: in the NL, ARN organises the collection of fuel (when not clean enough to be re-used), waste oils, brake fluid, coolant, screen wash and -since recently, 2004- air conditioning fluid. In Germany and the UK, the trend seems also to be the separation of several fractions.
- Practice B - separate storage of 2 fractions: waste oils + brake fluid, coolant + screen wash. In France, waste oil + brake fluid are burnt for energy recovery (e.g., in cement kilns – they are usually collected by collectors with a 0 or positive gate fee, i.e. the dismantler has to pay for having his waste oils treated). Coolant + screen wash can be sent for regeneration (with a positive gate fee in France).
- Practice C - separate storage of 1 fraction: in Hungary, waste oils + brake fluid are stored and sent for energy recovery or recycling. Coolant + screen wash are left in the body car and remain in the ASR.
- Practice D - no specific rule regarding the number of fractions to be stored separately, as in Poland.

Other informal practices exist. For instance, it is well known that on the ground, waste oils sent to regeneration are sometimes polluted by water-based fluids. When collectors collect for free without strong constraints and control, this is not likely to encourage dismantlers to separate the two types of fluids.

In uncontrolled dismantling places, waste oils and water-based fluids are partly spilt into the soil and partly left in the body car; they can then be found in oily waste at the shredding step. From interviews (in particular in Fr and G), it can not be excluded that part of the WO are drained out and sent to recovery; dismantlers motivation would be to avoid the working environment becoming too dirty (with an oily ground); the existence of free collection system in the country would facilitate this behaviour.

Regarding specific equipment (impermeable surfaces for appropriate areas with the provision of spillage collection facilities, decanters and cleanser-degreasers), as already mentioned, ATF not yet in line with the directive have not all made all the necessary investment yet.

c/ Practices prior to the Directive and main environmental problems associated

Fuels have always been separated from other fluids and re-used mainly on-site as already mentioned.

As mentioned here above, it can not be excluded that part of the WO were drained out and sent to specific facilities to avoid the working environment becoming too dirty.

But according to the interviews performed, the common practice of dismantlers regarding fluids other than fuels was to spill (part of) them into the soil and/or leave them in the body car. These fluids represent a main source of potential pollution from ELV as described hereafter, either due to spillage at the point of dismantling, inappropriate disposal or ASR contamination.

Environmental impacts linked to waste oils spillage

Example of average composition of waste oils

Light hydrocarbons (2-15%)
Heavy hydrocarbons (less than 80%)
Water (0-10%)
Additives, metals (0-10%)

Source: Total Company (http://www.total.com/static/en/medias/topic103/Total_2003_fs09_Used_lubricant_disposal.pdf)

The environmental impact of waste oils reversed in soil or water depends on the levels and types of contaminants present in the oil. The most toxic components of waste oils include heavy metals (arsenic, cadmium, chromium, etc.) and PAHs (Polycyclic Aromatic Hydrocarbons such as benzene, toluene, xylene). These highly toxic substances tend to concentrate in soil, water, and biota. Due to their high persistence in the environment and their tendency to bio-amplificate through food chains, they can accumulate directly or indirectly (through food chains) in humans causing adverse effects on human health. The latter include a wide range of illnesses, from irritations to cancer, anaemia, skin ulcerations and cardiovascular disease.

Animals and aquatic organisms will share some of the human health effects. Observed effects include acute toxicity[23] in aquatic organisms as a result of poisoning by heavy metals; acute toxicity in fish, and tumours, caused by mixtures of PAHs. Oil contaminants also have a range of properties poisonous to plants.

More detailed information can be found in New Zeland Ministry for the Environment official report (2000) 'Used Oil Recovery, Reuse and Dispoasal in New Zeland – Issues and Options', 55pp (http://www.mfe.govt.nz/publications/waste/used-oil-recovery-dec00.html).

23 There are two types of toxicity, acute and chronic. Acute toxicity refers to short term exposure to a toxin which produces symptoms within a short period of time after the exposure. Chronic toxicity is used to describe the potential long term effects which could result from exposure to a toxin over time.

Environmental impacts linked to waste oils improper incineration

When any substance is burned, the elements and compounds of which it is made up are released into the air as gases or particles, or they collect in the ash. If released in high enough quantities, some of these gases and particles can have harmful effects on human health[24] and the environment.

Used oil is not a homogeneous substance. Different oils may contain many different impurities. The use to which the original oil was put determines the types of contaminants contained in the used oil.

The combustion of oils containing carbon and chlorine can produce a wide range of organochlorine compounds. These can include 17 dioxins and furans, which pose a risk to human and environmental health. Toxic responses include skin toxicity, immunotoxicity, carcinogenicity, and adverse effects on reproduction, development and endocrine functions.

Some conditions are required to burn oil without causing adverse effects on human health and the environment, including: controlling the content of the substance burned, using filters and scrubbers to remove particles and chemicals from the discharge, designing chimney stacks to ensure good dispersion of the discharge, ensuring the burner operated to a particular degree of combustion efficiency, specifying methods of containing and disposing of ash.

More detailed information can be found in New Zeland Ministry for the Environment official report (2000) "Used Oil Recovery, Reuse and Dispoasal in New Zeland – Issues and Options", 55pp (http://www.mfe.govt.nz/publications/waste/used-oil-recovery-dec00.html)

Environmental impacts linked to water-based fluids spillage

Example of average composition of cooling liquid

Monoethylene glycol (30%) Water (60%) Additives (5%)

Source: LCA of electric vehicles compared to thermal vehicles, by BIO Intelligence Service for AVERE (Association pour le développement du véhicule électrique – Association for the development of electric vehicle), 2001

Once released into the environment, ethylene glycol partitions mainly into surface water or groundwater. It does not bioaccumulate or persist in the environment, primarily due to biodegradation. But as it biodegrades rapidly in the aquatic environment, it has the potential to induce depletion of the dissolved oxygen (DO) in receiving waters.

Laboratory tests exposing aquatic organisms to stream water receiving runoff from airports have demonstrated toxic effects and death. Terrestrial organisms are much

24 The effects on human health can be direct or indirect. Direct harm to human health can occur when the fine particles are inhaled into the lungs. Indirect effects occur when the fine particles, which contain contaminants such as heavy metals, settle on crops and end up in the food eaten.

less likely to be exposed to ethylene glycol and generally show low sensitivity to the compound.

However, available data from oral acute poisoning cases (humans) and repeated-dose toxicity studies (experimental animals) indicate that the kidney is a critical organ for the toxicity of ethylene. It also induces slight reproductive effects and developmental toxicity, including teratogenicity, namely in rodents exposed by the oral route.

The full environmental and human health risk assessments of ethylene glycol can be consulted on the International Programme for Chemical Safety INCHEM website (http://www.inchem.org/documents/cicads/cicads/cicad45.htm; http://www.inchem.org/documents/cicads/cicads/cicad_22.htm). These assessments have been published under the joint sponsorship of the United Nations Environment Programme, the International Labour Organisation, and the World Health Organization, and produced within the framework of the Inter-Organization Programme for the Sound Management of Chemicals; 2000.

Effects of having fluids remaining in the body car

Fluids contaminated ASR. Their presence in ASR was part of the motivation for the discussions about the classification of ASR as hazardous waste (in a context where they were generally landfilled as non hazardous waste).

d/ What if no ELV Directive

Without the Directive, fluids (except fuel as mentioned) would still be spilt into the environment or left in the body car (because there would still be a majority of illegal treatment installations and because it was a common practice whatsoever). And maybe part of the WO would be drained out and sent to specific facilities to avoid the working environment becoming too dirty.

Remark: from a theoretical point of view, one can say that due to the existence of the Directive 75/439/EC on Waste Oils (WO) amended in 1987, these practices would have been illegal and thus that operators would have been obliged to separate fluids and send them to required treatment plants. But would the Waste Oils Directive alone have been able to reach this result?
First a majority of illegal dismantlers would still exist without the ELV directive thus not complying with the WO directive either.
Another useful information concern the implementation of the WO directive itself. As shown in the TN SOFRES Consulting & BIO Intelligence Service study in 2001[25], more than 20 years after the implementation of the WO directive, about 25% of WO were still illegally disposed of in 1999 (spillage in the environment or combustion of unprocessed used oil as fuel)[26]. Because dismantlers were largely illegal, it is likely that fluids from

25 'Critical Review of Existing Studies and Life Cycle Analysis on the Regeneration and Incineration of WO', by TN SOFRES Consulting & BIO Intelligence Service, for the EC – DG ENV, in 2001: this is an extended impact assessment of different options in the framework of the revision of the WO directive (http://europa.eu.int/comm/environment/waste/studies/oil/waste_oil.htm)

26 The remaining 75% were disposed of as follows: an average of 25% of the collectable WO (and 33% of the collected WO) are believed to have entered a regeneration plant in Eu15 in 1999; about 50% of WO were energetically used (of which an average of 35% in cement kilns, with large discrepancies

ELV were among these 25% of WO illegally disposed of and that without the ELV directive, this situation would not have changed.

e/ Potential benefits in case of full compliance with the ELV Directive

Local/regional benefits linked to fluids spillage avoided

The following **potential risks** are **avoided**:

- **Contamination of soil, water, and biota[27] by potentially 66 kt of oils per year**, in particular by heavy metals and PAHs which can cause adverse effects on **human health** by direct exposure (ingestion, inhalation, dermal absorption) or indirectly through food chains (by eating contaminated organisms). This may lead to a wide range of illnesses, from irritations to cancer, anaemia, skin ulcerations and cardiovascular disease), **acute toxicity in aquatic organisms and fish and poisoning of plants**.
- **Contamination of aquatic environment, mainly** surface waters and groundwaters **by potentially 54 kt of water-based fluids per year**, in particular by ethylene glycol which can induce depletion of the dissolved oxygen in receiving waters **causing toxic effects and death of aquatic animals**; terrestrial organisms and humans generally show low sensitivity to the compound, but adverse effects (kidney dysfunctions, reproductive and developmental toxicity) have been observed in cases of oral poisoning.

Global benefits linked to proper management of waste oils

The results presented below are extracted from the last reference study about WO known to us: the above mentioned 2001 TN SOFRES Consulting & BIO Intelligence Service study performed for DG ENV. They are based on a critical assessment of the four Life Cycle Analysis (LCA) studies available at the time for WO management.

They cover the 5 following technologies existing in Europe, three regeneration technologies which can be considered being representative of a diversity of regeneration technologies, including modern processes, and two incineration options:

- Vacuum distillation + clay treatment,
- Vacuum distillation + chemical treatment,
- Hydrogen pre-treatment + vacuum distillation.
- Incineration in cement kiln,
- Incineration in asphalt plant.

In order to be able to compare these 5 treatment options, the following systems were studied.

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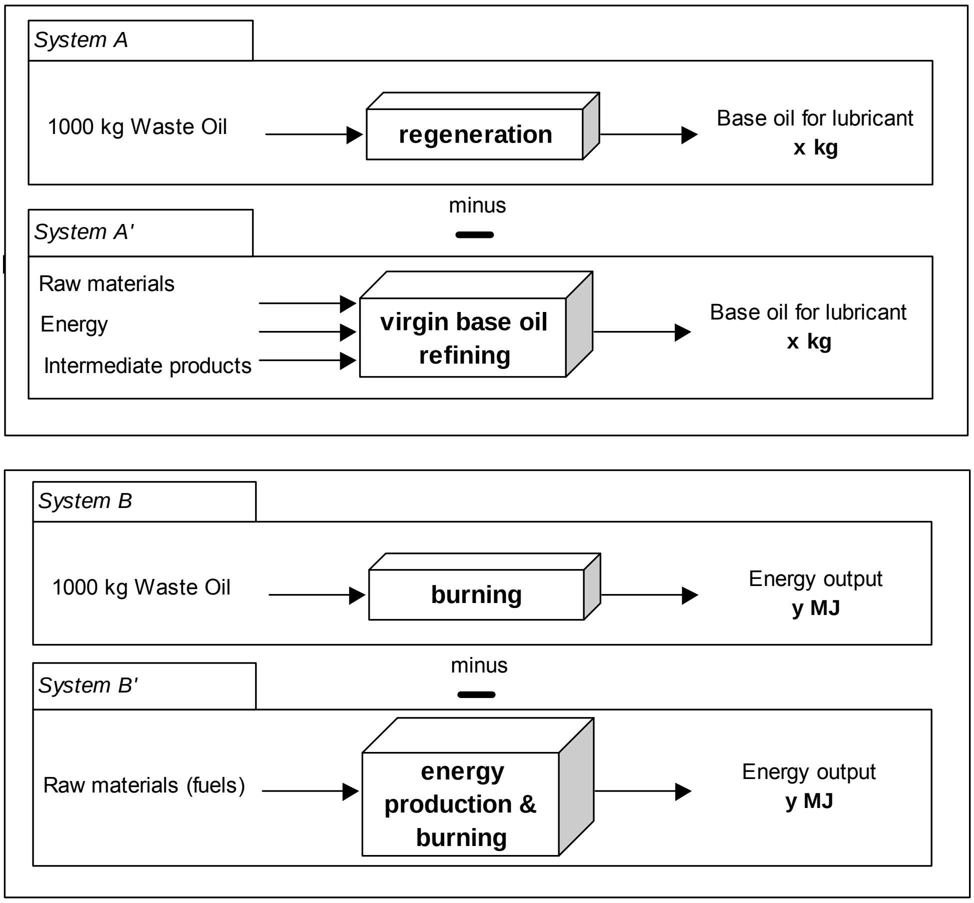
within MSs). Whereas the WO directive gives priority to the regeneration of WO over their incineration, Member States do indeed not favour regeneration of WO; on the contrary they are widely using WO as fuel in industrial installations. But this is another issue, and a reason why the WO directive is presently under discussion.

27 living organisms found in a given area

Figure 4.1: System boundaries for regeneration options



Remark: System A describes two functions (or objectives): 1°) the disposal of 1000 kg WO, and 2°) the production of x kg of base oil. System A' describes one function: the production of x kg of base oil. Thus, the overall system (A-A') was designed to reduce the system to only one function: the disposal of 1000 kg WO. Idem for system (B-B'). Having the same function, (A-A') and (B-B') can thus be compared (from an environmental point of view).

Four environmental indicators were analysed:

- Consumption of fossil energy resources
- Contribution to global climate change
- Contribution to regional acidifying potential
- Emission of Volatile Organic Compounds (VOC)

A qualitative analysis was performed due to the lack of comparable quantitative data.

Remark: these studies are LCAs, not risk assessments. They assess potential global impacts (as in any LCA - see section 8.1.3), not real local impacts. No assessment available about local risks specific to WO regeneration or incineration facilities was identified in the framework of this study.

The following conclusions drawn from the LCAs analysed were those considered sound:

- The environmental burden of the recovery treatment (regeneration or incineration) by itself is generally less important than the one of the avoided process (virgin base oil production or traditional fuel or energy production).
- Within a life cycle perspective, the total contribution of the management system under consideration is indeed the result of the difference between two different quantities: the impact of the recovery treatment minus the impact of the main avoided system (this latter representing a bonus). The environmental impacts of WO recovery systems are mainly determined by this bonus and less by the direct impacts of the recovery processes themselves.
- The 5 WO recovery options under consideration are favourable in terms of environmental impacts (i.e. they contribute to avoid impacts) by comparison with a 'do nothing' system.
- The amount of the bonus brought by the avoided process is determined by the choice of the substituted process (this is also the case for other wastes with a high calorific value as plastic wastes).
- Especially in the case of the incineration of WO with energy recovery, the type of fuels that WO replace is crucial: fossil fuel, hydroelectricity, thermal electricity, other wastes….

Of course these favourable conclusions can not be extended to all types of regeneration or incineration processes (in particular to old plants or to all types of substitution). And also the following issues have not been addressed in the LCAs available and can be considered as gaps: noise, odour, nature conservation (biodiversity, etc.), land use, toxic emissions (heavy metals, dioxins…).

But in the framework of this ELV study, the important result is that: **WO recovery options exist which are beneficial to the environment** (by comparison with a 'do nothing' situation where base oil or energy is produced from raw materials), i.e. **WO recovery does not only avoid local environmental impacts (those linked to their spilt in the environment) but can also create environmental bonus** (if properly managed).

Remark: in order to grasp what quantities are at stake, let's calculate the number of km that a car could make if all the energy contained in the 66 kt of WO arising in Europe were recovered.
Considering an ICV (inferior calorific value) of 39.3 MJ/kg of waste oils, 2.6×10^9 MJ are contained in these 66 kt [$66 \times 39.3 \times 10^6 = 2.6 \times 10^9$].
Considering an average vehicle consumption of 8 litres of gasoline per 100 km (with a density of 0.75 kg/l and an ICV of 42.8 MJ/kg), these 2.4×10^9 MJ of energy would allow a car to make 950×10^6 km [8 x 0.75 x 42.8 / 100 = 2.568 MJ/km; 2.6×10^9 / 2.568 = 1010×10^6) which correspond to **a car turning around the earth 25,250 times**[28].

Other effect of the Directive

There has been discussion over the years about ASR being classified as hazardous waste. The removal of fluids before shredding involves that ASR will be free from

[28] Circumference of the earth = 40,000 km

waste oils coming from ELV which are hazardous waste. According to stakeholders interviewed, this (together with the removal of batteries and thus lead) is contributing to render his discussion less relevant (for some MSs at least). However, shredders do not treat ELV only; they treat all kinds of other waste (such as WEEE, ferrous and non ferrous pieces) which can bring contaminants to ASR.

If ASR were to be declared non hazardous waste, the effect would be more availability of hazardous waste landfills (or rather no necessity to create new hazardous waste landfills to absorb them; as a matter of fact, according to stakeholders interviewed, the current situation is that there is an important gap between available capacity in hazardous waste landfills and ASR produced in Europe, i.e. an under-capacity).

f/ Benefits to-date due to the ELV Directive

They are not easy to quantify as even if 50% of ELV are now treated in ATF due to the ELV directive, ATF do not all comply with the directive yet.

But if we anticipate their compliance and thus assume that 50% of ELV being treated in ATF in line with the directive, then the implementation of the Directive to-date avoids having about half of these fluids spilt in the environment or left in ASR and instead make them be transferred to appropriate facilities for recovery.

The benefits to-date are similar to the local and global benefits described here above in case of full compliance with the directive except that they concern only about half[29] of the fluids quantities:

- 31 kt of waste oils per year (instead of 66 kt)
- 25 kt of water-based fluids per year (instead of 54 kt)

29 See the box at the beginning of this section dealing with 'Topic 2 - Depollution of ELV: fluids': about 5 millions vehicles out of 10.8 millions.

4.7 Topic 3 - Depollution of ELV: batteries

Key figures

	Example of composition		Total in ELV requiring treatment	Total in ELV treated in ATF
Lead containing components	8,6 kg	*64%*	93 kt in Eu25/yr	43 kt in Eu25/yr
Electrolyte (sulphuric acid)	3,8 kg	*28%*	41 kt in Eu25/yr	19 kt in Eu25/yr
Polypropylene	0,7 kg	*5%*	8 kt in Eu25/yr	4 kt in Eu25/yr
Other (separator…)	0,4 kg	*3%*	4 kt in Eu25/yr	2 kt in Eu25/yr
Total	**13,5 kg**	*100%*	**146 kt in Eu25/yr**	**68 kt in Eu25/yr**
	(1)		(2)	(3)

(1) Source: 'The Environmental Impacts of Motor manufacturing and Disposal of ELV: Moving Towards Sustainability', DTI (UK), 2000; similar data for a modern battery (i.e. in a PP-casing) in www.gtz.de/de/dokumente/en-recycling-of-batteries.pdf, 2002

(2) 10.8 millions ELV to be treated in Eu25 in 2005

(3) 4-6 millions ELV treated in ATF in 2005; 5 millions considered for the calculations

a/ Requirements of the Directive

Annex 1 of the Directive requires:

- Removal of batteries and appropriate containers for storage
- Appropriate storage tanks for battery acids (electrolyte)
- Electrolyte neutralisation on site or elsewhere

b/ Current practices

Spent starter batteries in general

Spent starter batteries are constituted of two different flows:

- OEM and after market batteries[30] when they are spent,
- spent batteries contained in ELVs.

80-95%[31] of spent starter batteries available for collection are believed to be collected and sent to recycling. No statistics exist at the EU level to confirm this situation. But returning used lead batteries (starter batteries as well as industrial batteries) to the recycling loop has a long tradition. Thanks to the compactness of a battery, its high lead proportion and relatively high metal prices, it has been worthwhile for last owners to return old batteries to the scrap trade or secondary smelters. The return rate of spent batteries was thus already high in times when resource conservation and environmental protection, recycling, etc. did not yet play a role.

30 Original Equipment Manufacturer's batteries i.e. those sold in cars; when they are spent, they are replaced by AM (After Market) batteries

31 Source: 'IEA Batteries Impact Assessment on Selected Policy Options for Revision of the Battery Directive' carried out by BIO Intelligence Service for the EC DG ENV in 2003 (see pages 7 and 38) - http://europa.eu.int/comm/environment/waste/batteries/pdf/eia_batteries_final.pdf

Approximate composition of lead-bearing components of a starter battery

Grid metal, poles, bridges		44%
Pb	96-98%	
Sb	2-4%	
Ca	<0,5%	
Paste		56%
$PbSO_4$	60%	
PbO (PbO_2)	19%	
PB	21%	
Total		100%

Source: Fundamentals of the Recycling of Lead-Acid Batteries, GTZ, 2002 (www.gtz.de/de/dokumente/en-recycling-of-batteries.pdf)

Spent lead batteries are recycled both in industrial facilities and by informal small enterprises. As described by GTZ[32], industrial recycling smelters use both the grid metal and the lead-containing paste to produce secondary lead. The informal sector, in contrast, often only uses metallic parts to produce articles such as weights for fishing nets; the other parts of the batteries are dumped in the environment.

Regarding the electrolyte neutralisation in these industrial facilities, used batteries are emptied by hand and used acid is collected in plastic barrels where it is purified by sedimentation and decantation. Purified acid is packed for sale. Possible customers are the mining and metallurgical industry which uses acids in various leaching operations.

Spent starter batteries from ELV

In the absence of available statistics, what can be said about the collection and recycling rate when focusing only on spent batteries contained in ELVs (spent batteries coming from scrapped ELV were assessed representing about 15% of the total spent starter batteries arising in Europe[33 34])?

Evidence from interviews suggests that there are different practices by dismantlers:

- Practice A: whole batteries (including the electrolyte) are removed and sold to dedicated treatment facilities
- Practice B: the battery is left in the body car which is sent to the shredder
- Practice C: the battery is mashed open to remove and sell the lead-bearing components; the electrolyte spills in the soil and the remaining waste is dumped in the environment.
- Practice D: batteries are removed and sold, without the electrolyte, to dedicated treatment facilities; the electrolyte is drained (probably with a view to avoiding

32 www.gtz.de/de/dokumente/en-recycling-of-batteries.pdf, *2002*

33 *Source: 'IEA Batteries Impact Assessment on Selected Policy Options for Revision of the Battery Directive' carried out by BIO Intelligence Service for the EC DG ENV in 2003 (see page 7) -* http://europa.eu.int/comm/environment/waste/batteries/pdf/eia_batteries_final.pdf

34 based on the assumption that about 45% of ELV were scrapped, most of the remaining ones being exported

risk of fire from batteries and then facilitating their handling) and spilt in the environment (to avoid any storage constraint)

All stakeholders interviewed affirm that the common practice in their country has been practice A for a long time (the NL mentioned that it concerns about 90% of batteries). Today, practice A is claimed to be the one of ATF in line with the ELV directive. Segregated batteries are then sent to industrial facilities with high pollution control standards.

Practices B, C and D are typical of illegal dismantling facilities and may also exist in ATF not yet complying with the ELV directive.

Remark regarding practice B (batteries left in the body car):

This practice does not indeed seem absent from ATF.

The return rate is actually determined largely by the potential earnings of scrap collectors and traders. In case of low lead prices (e.g., over the 1990-1993 and 1996-2000 periods), scrap dealers have no financial motivation to take batteries out. As reminded by Ernst&Young[35], one reason for leaving the battery into the scrap is well known: when the decrease of the battery value is concomitant to an increase of the scrap value, then the interest for scrap dealers to separate batteries from the rest of the scrap disappears with, as a direct effect, an increase of the quantity of batteries contained in scrap (phenomenon measured by the presence of lead in scrap residues).

It is likely that statutory requirements or certification standards existing in MSs to take back spent batteries have compensated, at least in the cases of ATF in line with the ELV directive, for the loss of economic incentives in spent batteries return.

But anecdotal evidence from stakeholders interviewed during this study (dismantlers themselves) suggested that the value of car batteries might have declined over the last 2 years again to the point where proper disposal was less economically viable. In such cases, batteries are either left in the body car or dumped in the environment.

c/ Main practices prior to the Directive and main environmental problems associated

On one hand, as above mentioned, all stakeholders interviewed affirm that the common practice in their country has been the removal and recycling of whole batteries for a long time.

On the other hand, the ELV dismantling sector being dominated by illegal operators in the past, it could be reasonable to consider that a significant proportion of batteries from ELV were not properly treated and assume that practices B, C and D were common, i.e. in particular electrolyte left in the body car with the battery or spilt in the environment. Did this concern ‘only’ 10-15% of spent batteries or more? We are not able to conclude from available information.

35 ‘Étude économique sur la filière de recyclage des véhicules hors d’usage’ (Economic Study on the Management of ELV), Ernst&Young, for French Environment Agency Ademe, Sept 2003

Besides, with the Batteries directive (91/157/EEC), industrial recycling smelters with high pollution control standards had developed. But, in the absence of statistics here again, it is quite reasonable to consider that due to the high proportion of illegal dismantlers prior to the ELV directive, the informal sector for batteries recycling was favoured.

The potential health and environmental risk involved when processing battery scrap is very high as described hereafter.

Environmental impacts linked to the electrolyte spillage

Sulphuric acid discharged in the environment poses substantial health risk to aquatic organisms and soil fauna mainly due to its corrosive and irritant properties and its capacity to rapidly cause substantial changes in the pH of soil and/or water. Laboratory and field studies show that even at very low concentrations, this acid is particularly toxic to aquatic ecosystems, namely to fish and algae.

Since the soil mobility of sulphuric acid is very high, once it enters the soil, it can readily reach groundwater or surface waters and endanger drinking-water supplies.

In all biota, including humans, the contact with sulphuric acid causes severe burnings. Moreover, according to the International Agency for Research into Cancer (IARC), the occupational exposure to strong inorganic acid mists containing sulfuric acid is carcinogenic.

More detailed information on toxicity and ecotoxicity of sulphuric acid from the International Uniform Chemical Information Database can be found on the European Chemicals Bureau website (http://ecb.jrc.it/). The full dossier is a compilation based on data reported by the European Chemicals Industry and it contains all (non-confidential) information from the single datasets, submitted to the IUCLID/HEDSET – 2000; 179 pp.

Other environmental impacts linked to improper batteries treatment

According to GTZ, depending on the level of mechanisation and environmental standards, the following hazards can arise:

- Wind dispersal of lead dust if crushed battery scrap is stored without protection
- Substantial atmospheric emissions (lead-containing dust, soot, SO, chlorides, dioxins, etc.) when battery scrap is melted (e.g. in illegal scrap yards or uncontrolled burning equipment) due to: processing the entire battery including its organic parts (PP casing for instance), inadequate removal of gases and vapours during the smelting and refining process, absent or inadequate flue gas treatment
- Open tipping of residues and waste such as batteries casings

d/ What if no ELV Directive

Batteries management would be market-driven as it has always been for starter batteries, with lots of informal and illegal practices as just described.

It is likely that the Batteries directive alone would not have changed behaviours as it contains no collection targets for starter batteries and no clear responsibilities (whereas in the ELV directive, dismantlers are the ones obliged to remove batteries for a proper treatment).

e/ Potential benefits in case of full compliance with the ELV Directive

If we consider, as many stakeholders put it during our interviews, that **the main benefit of the directive is to capture** the **10-15% of spent batteries** which would have been improperly treated otherwise, the benefits are then as follows.

Local/regional benefits linked to electrolyte spillage avoided

The **potential risk of contamination of water** (directly or from soil) **by 4 to 6 kt of sulphuric acid per year** is **avoided** (toxicity to aquatic life, contamination of water supplies).

Local environmental benefits linked to proper batteries treatment

Assuming that the 93 kt of lead-containing components contain an average of 50% of lead, an additional of about 4 to 6 kt of lead would be diverted from waste (10-15% of 93 kt x 50%).

Besides, as already mentioned, compared to low standard installations, controlled ones limit (or avoid):

- Wind dispersal of lead dust
- Substantial atmospheric emissions (lead-containing dust, soot, SO_2, chlorides, dioxins, etc.)
- Open tipping of residues and waste such as batteries casings

Other local and global impacts/benefits linked to proper management of batteries

Starter batteries are recycled in lead smelting plants, located in most of European countries. About 0.58 t of lead is recovered from 1 tonne of battery smelted (58% recovery rate).

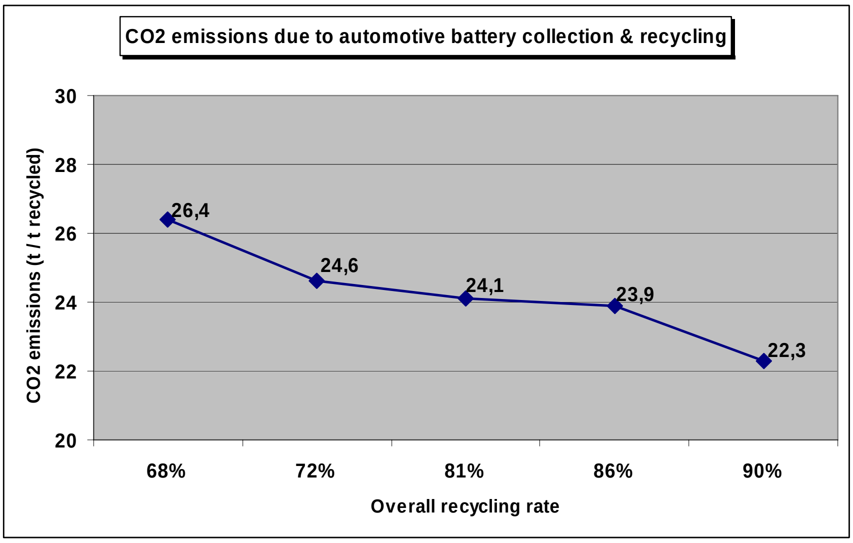
The results presented below are extracted from the EIA carried out by BIO Intelligence Service about the Batteries directive[36] with primary data from an ERM study[37] (which simulated environmental impacts of different lead-acid automotive battery collection and recycling scenarios in UK using a life cycle assessment (LCA) approach).

- The higher the collection and recycling targets, the higher the lead diverted from waste
- **There are negative consequences of recycling**: environmental damages linked to collection, transport and re-processing (in particular to air) are higher than benefits brought by virgin material savings

36 *'IEA Batteries Impact Assessment on Selected Policy Options for Revision of the Battery Directive' carried out by BIO Intelligence Service for the EC DG ENV, 2003 (see pages 18, 84 and 87)* *http://europa.eu.int/comm/environment/waste/batteries/pdf/eia_batteries_final.pdf*

37 '*Analysis of the environmental impact and financial costs of a possible new European directive on batteries*', 2000

- **But the negative consequences of recycling decrease when recycling rate increases** (for a given collection target, the higher recycling target, the lower negative consequences of recycling: recycling benefits increase more than transport negative impacts)



Source: http://europa.eu.int/comm/environment/waste/batteries/pdf/eia_batteries_final.pdf (page 84)

Results indicate a clear environmental interest in the scenarios with higher overall recycling rates. With respect to the other environmental indicators, a similar presentation would have shown a similar trend.

As a conclusion, from available literature, **batteries recycling generate global environmental impacts (in particular to air). But the higher the recycling rate, the lower the impacts**. And it also diverts lead from waste.

Another interesting results highlighted in the EIA carried out by BIO Intelligence Service[38] is the following: the rate of re-use of secondary lead in new batteries is of major importance for the environmental impact; and **the larger quantity of recycled lead in a lead-acid battery, the less environmental damages of its life cycle**.

Remark: because batteries recycling generates environmental impacts, one question comes up: what are the 'acceptable' alternatives and do they generate more environmental impacts than recycling? 'Acceptable' alternative is for instance authorized hazardous waste landfill. Alternatives to recycling were not among the options selected by the EC and analysed in the EIA performed by BIO IS. Results from an Environmental Risk Assessment would be necessary to answer this question. But this analysis would have to cover the production of batteries in order to have a whole picture and not forget that the use of recycled lead in a lead-acid battery has benefits for the environment.

Other effect of the Directive

[38] and based on data presented in the following publication: *Environmental assessment of vanadium redox and lead-acid batteries for stationary energy storage*, C.J. Rydh, Journal of Power Sources 80 (1999) 21-29

There has been discussion over the years about ASR being classified as hazardous waste. The separation of batteries before shredding involves that ASR will be free from lead sulfate which would have otherwise formed from the electrolyte. According to stakeholders interviewed, this (together with the removal of fluids as already mentioned) is contributing to render this discussion less relevant (for some MSs at least).

f/ Benefits to-date due to the Directive

As for fluids, they are not easy to quantify as even if 50% of ELV are now treated in ATF due to the ELV directive, ATF do not all comply with the directive yet.

And also it is not possible to conclude if part of the 10-15% of spent batteries to be captured thanks to the implementation of the directive have already been captured in the ATF or if they are those contained in the other 50% of ELV not yet treated in ATF.

But if we assume that the directive to-date already allowed capturing 50% of the remaining 10-15% of spent batteries not properly treated, then the environmental impacts and benefits to-date are similar to the local and global ones described here above in case of full compliance with the directive except that they concern only about half of the quantities. To date, the directive would then ensures that:

- 2 to 3 kt of sulphuric acid per year are not spilt in the environment
- about 2 to 3 kt of lead are diverted from waste
- 500 to 750,000 batteries are recycled in controlled installations (10-15% of 10.8 ELV x 50%)

4.8 Topic 4 - Depollution of ELV: liquified gas tanks

Key figures

0.06 kg/ELV[39] % of ELV containing liquefied gas tank: very low in most MSs (the highest would be in the NL, with about 8% of ELV[40])

a/ Requirements of the Directive

Annex 1 of the Directive requires from treatment operations for depollution of end-of-life vehicles:

- The removal of liquefied gas tanks
- Removal or neutralisation of potential explosive components

b/ Current practices

Liquefied gas tanks (LGT) are relatively new components for dismantlers / shredders as they concern modern vehicles amongst which not so many are at the end of their life yet.

According to ARN, about 8% of ELV arising in the NL contain a LGT, composed of a steel tank containing propane. In the other MSs where interviews were performed, evidence suggests that the presence of LGT in ELV treated is still very rare.

In the NL, dismantlers having contracted with ARN are obliged to remove the tanks and send them to certified de-gazing companies where propane is either sold or burnt. They then have the possibility to take back empty tanks to either sell them as second-hand parts or deliver them to steel recycling facilities. ARN started this practice in 1996 under the pressure of shredders when the proportion of LGT reached a level which began to create a significant risk of explosion during the shredding process.

In the other MSs, several practices co-exist at the dismantling place:

- Practice A: LGT removed and sent back to the producer
- Practice B: LGT removed, propane neutralised on-site (in flare) and tank put back in the body car
- Practice C: LGT removed and stored waiting for larger quantities to be treated
- Practice D: LGT left in the body car to explode during the shredding process

For instance in France, shredders have alerted the Dismantlers Federation about the problem linked to the explosion. The Federation has decided to update the certification standard by including the obligation for dismantlers to remove LGT. Practices A, B and C are now common to certified ATF in France.

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39 Source: ARN (NL), Jan 2006 (http://www.arn.nl/engels/3resultaten/33.php)

c/ Practices prior to the Directive and main environmental problems associated

Except in the NL where the removal has existed for 10 years, in the other countries, fewer LGT than today were present and common practice was to leave them with the body car to be shredded where they would explode.

Given that LGT quantities were still very low in Europe at the time of the elaboration of the directive, one can assume that the directive wanted not primarily to address an existing environmental problem but rather to anticipate a problem which would arise when more LGT would have been present in ELV.

These potential environmental problems include:

- Uncontrolled propane emissions into the air. Propane emissions have an impact on photochemical oxidation meaning that their presence in the air can lead to the creation of the ozone that can cause skin and eye irritations.
- Noise due to the explosion? (only one of the stakeholders interviewed mentioned the noise as a possible nuisance for residential population).

But the main potential problem mentioned by stakeholders interviewed is the security for workers during the shredding,

d/ What if no ELV Directive

From experience in the NL and more recently in France, one can make the case that under the pressure of shredders, above a certain proportion of LGT in ELV (which can not be established from available information except that it is probably near or below 8%), dismantlers would have decided to remove LGT.

As informal dismantling facilities would dominate the sector, one can also make the case that whole LGT would have been dumped in the environment or propane released on-site and tanks sent for steel recycling.

Before this proportion of LGT in ELV be reached (the one which makes LGT unacceptable to shredders anymore), one can assume that most dismantlers would have leaved them with the body car to be shredded.

e/ Potential benefits in case of full compliance with the ELV Directive

The main benefit of the directive resides in the fact that LGT are removed (even if in small proportion in ELV) and thus avoids the risk of explosion during shredding and that propane is neutralised (rather than being released into the air).

f/ Benefits to-date due to the Directive

Benefits to-date concern the progressive change in behaviour, not in the NL where good practices are prior to the directive, but in the other countries where ATF in line with the directive begin to remove LGT. However practices are not stabilised yet given that concerned quantities are still low.

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40 Source: ARN (NL), Dec 2005

4.9 Topic 5 - Depollution of ELV: air bags

Key figures

% of ELV containing a working air bag: n.a.

a/ Requirements of the Directive

Annex 1 of the Directive requires from treatment operations for depollution of end-of-life vehicles the removal or neutralisation of potential explosive components (e.g., air bags).

b/ Current practices

Air bags do not seem to cause a problem today yet as the proportion of vehicles with working air bags is still low according to stakeholders interviewed (no statistics available).

- First, air bags are found only in (some of the) most recent vehicles (those put on the market during approximately the last 8 years). According to a web article[41], about 10% of vehicles arising at scrapyards would now have air bags.
- Secondly, one has to distinguish between working air bags (which have the potential to explode in shredders) and the others which have been damaged or have detonated before arriving at the dismantling facility (for instance during a car accident).

Several practices co-exist in the countries analysed:

- Practice A: dismantlers remove working air bags and sell them for re-use (secondary air bags)
- Practice B: air bags are removed by dismantlers and deployed in a separate chamber
- Practice C: air bags are blown up, by dismantlers, inside the ELV placed in a controlled environment
- Practice D: air bags are left within the body car and, for the working ones, explode in the shredding equipment

Practice A is still the common practice according to stakeholders interviewed (even in the NL; ARN is working on a recycling solution expected to be ready when necessary, i.e. within 5 years or so; they indeed anticipate that this is in 5 years that the number of ELV with air bags will reach a level requiring a new solution).

Practice D also exists and is not claimed to cause a problem to shredders to date, considering the low proportion of body cars concerned.

Practices B and C are new (for instance they begin to develop in France in certified ATF and in the UK).

41 http://www.letsrecycle.com/news/archive/news.jsp?story=4888

In illegal facilities, air bags are either removed to be sold or left in the car to explode during the shredding.

Remark 1: operators and other stakeholders are debating about the practices that should be authorised or not. For instance, certain would prefer that the re-use of air bags be forbidden or at least limited. Others refer to a US study which would conclude that their re-use should be accepted (one of the argument being that this is common practice amongst car repairers: when they take out a working air bag to repair a car, they put the same air bag back; if they had to put a new one, they would have to ask the car owner to pay for about 1,000 Euros).

Some want to consider that the explosion occurring during shredding is a form of neutralisation.

Remark 2: new vehicles put on the market today contain air bags designed with a neutralisation system (to make them explode on command). This is not the case with vehicles produced 15 years ago and treated now.

c/ Practices prior to the Directive and main environmental problems associated

Working air bags were either removed to be sold or left in the car to explode during the shredding.

Regarding the potential environmental problems associated, no literature was identified in the course of the study dealing with this aspect, except a web article[42] which mentions that 'some of the airbags contain sodium azide, which is dangerous – one driver's airbag would have enough potentially to kill 35 people'. But no other evidence was identified to confirm / precise the importance of this risk.

The only potential problem mentioned by stakeholders interviewed is the security for workers during the shredding,

d/ What if no ELV Directive

It seems reasonable to stakeholders interviewed to make the case that practices would be mostly similar to what they are today since the directive did not change behaviours yet (proportion of ELV with air bags being still low).

e/ Potential benefits in case of full compliance with the ELV Directive

Local benefits

No more working air bags left in the body car thus no more potential worker security risk at the shredding place.

f/ Benefits to-date due to the Directive

No major environmental benefits to date can be attributed to the directive since practices did not evolve significantly yet (proportion of ELV with air bags being still

42 http://www.letsrecycle.com/news/archive/news.jsp?story=4888

low), **except that solutions are developing on the ground** to remove and neutralise air bags (this will generate more benefits within 5 years or so, when air bags in ELV are more numerous).

4.10 Topic 6 - Removal of pieces: catalysts

Key figures

		Total in ELV requiring treatment	Total in ELV treated in ATF
Ceramic-based catalysts			
ceramic material	900 g/ELV		
of which precious metals			
platinum	*1,2 g/ELV*	*13 t in Eu25/yr*	*6 t in Eu25/yr*
palladium	*0,176 g/ELV*	*2 t in Eu25/yr*	*0,9 t in Eu25/yr*
rhodium	*0,274 g/ELV*	*3 t in Eu25/yr*	*1,4 t in Eu25/yr*
	(1)	(2)	(3)

(1) EGARA, December 2005 (interview)
(2) 10.8 millions ELV to be treated in Eu25 in 2005
(3) 4-6 millions ELV treated in ATF in 2005; 5 millions considered for the calculations

a/ Requirements of the Directive

In order to promote recycling, Annex 1 of the Directive requires from treatment operations the removal of catalysts.

b/ Current practices

Ceramic-based catalysts have a significant value on the recycling market (up to 50 Euros / piece according to EGARA) as they contain about 2 g of precious metals[43]. Dismantlers (legal or illegal) are thus motivated to send them for recycling.

The removal of precious metals is a highly complicated process. For that reason, all catalysts, even those from illegal facilities, end up in industrial facilities (about 10 plants in Europe). The refining is then performed in the US who have a 25-year experience of using catalysts and recycling platinum (in particular in new catalysts).

c/ Practices prior to the Directive

No major change in behaviours can be identified.

d/ What if no ELV Directive

Catalysts would be mostly separated and sold for there precious metals to be recovered as today.

e/ Benefits to-date due to the Directive

Stakeholders interviewed agreed to say that the ELV directive has not involved major change in practices by dismantlers.

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43 Another type of catalysts exists: steel-based catalysts, containing about 5 kg of steel (based on a price of 120 Euros / t of steel, they have a much lower value than ceramic-based catalysts: 0,6 Euros / piece).

4.11 Topic 7 - Removal of pieces: tyres

Key figures

22-30 kg / ELV

Preliminary remark: the issue of tyres is a big issue in itself which goes far beyond the present study. As a matter of fact, there is no Directive but national legislations have been existing for several years or are developing, with landfill ban (e.g., UK) and/or producer responsibility (e.g., Fr). Lots of resources would be necessary to establish a precise overview of the situation in Europe. The fact that the Landfill directive (1999/31/EC) places a ban on the landfilling of whole used tyres by 16 July 2003 and shredded used tyres by 16 July 2006[44] is likely to have been a strong driver in behavioural change. Only partial information is provided hereafter.

a/ Requirements of the Directive

In order to promote recycling, Annex 1 of the Directive requires from treatment operations:

- Removal of tyres if these materials are not segregated in the shredding process in such a way that they can be effectively recycled as materials
- Also appropriate storage for used tyres, including the prevention of fire hazards and excessive stockpiling

b/ Current practices

Tyres in general

Tyres have traditionally been landfilled in Europe. Today, a wide variety of practices exist in Europe. They include:

- Re-use of newer tyres, subject to legal standards on tread
- Re-use for landfill engineering (whole tyres can be used in construction of landfill sites)
- Recycling through re-treading
- Recycling through grinding (crumb is used in sports and play surfaces, brake linings, landscaping mulch, carpet underlay, absorbents for wastes and shoe soles, and in rubberised asphalt for roads; some crumb is also used in tyre manufacture, along with virgin rubber)
- Other recycling techniques include cryogenic fragmentation, de-vulcanisation, microwave technology, and are subject to continuing development
- Energy recovery (through burning, pyrolysis, or incineration in cement kilns for instance)

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44 This ban does not include tyres used in landfills as engineering material.

Tyres from ELV

The NL established an obligation for dismantlers to remove tyres. In the other countries covered, the 2 options are still possible and indeed exist on the ground (in accordance with the directive): removal by dismantlers or segregation by shredders.

None of these countries has opted for recycling only (contrary to the requirement of the ELV directive).

In illegal facilities, one can expect that good tyres (an average of 2 out of 5 per ELV according to CNPA, Fr) are removed by dismantlers and sold for re-use and that the other tyres are either stockpiled or left with the body car and landfilled with ASR.

c/ Practices prior to the Directive and main environmental problems associated

Tyres were mainly re-used (for the good ones), landfilled or stockpiled.

According to the Ministry of Environment in Austria[45], used tyres present a difficult management problem in landfill or when stockpiled because of their volume, their resource loss and the fire hazard they pose.

d/ What if no ELV Directive

Would the same evolutions as those observed in MSs have occurred due to the Landfill directive and the national legislations being implemented?

Stakeholders interviewed had different view on this question: some of them (in Fr, G, UK) considered that the Landfill directive was more important as a driver while some others (in the NL and Pl for instance) consider the ELV directive as a real incentive (in the fact that it contains clear obligation upon specific actors –namely dismantlers and shredders- to remove/segregate).

e/ Potential benefits in case of full compliance with the ELV Directive

One can probably make the case that, when all ATF comply with the ELV directive and if 100% of ELV are transferred to ATF, the ELV directive will have contributed (along with the Landfill directive) to the disappearance of tyres landfilling and stockpiling practices, by establishing clear responsibility upon specific actors (dismantlers and shredders).

Regarding their recycling, it is likely that recycling and energy recovery practices will keep co-existing for a while (waste burning with energy recovery in cement kiln represents today about only 12% of total cement kiln capacity available in Europe for waste burning[46] - there are still lots of capacity to absorb tyres and other wastes).

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45 ‘Environmental Impacts of ELV – An information paper’ , Department of the Environment and Heritage, 2002 (http://www.deh.gov.au/settlements/publications/waste/elv/impact-2002/)

46 Confidential source of information from industry.

f/ Benefits to-date due to the Directive

One can also make the case that, the ELV directive **contributed (together with the Landfill directive) to the disappearance of tyres landfilling and stockpiling practices for ATF in line with the directive**.

4.12 Topic 8 - Removal of pieces: glass

Key figures

		Total in ELV requiring treatment
Windshield Back windows	15,2 kg/ELV	164 kt in Eu25/yr
Side windows	6 kg/ELV	65 kt in Eu25/yr
	(1)	(2)

(1) INDRA (dismantlers, Fr), Nov 2005
(2) 10.8 millions ELV to be treated in Eu25 in 2005

a/ Requirements of the Directive

In order to promote recycling, Annex 1 of the Directive requires from treatment operations the removal of glass.

b/ Current practices

Only in few MSs dismantlers are obliged (e.g., Dk, Pl by the law) or have incentive (Sw, NL, through the certification standard or financial support) to remove glass from ELV and send it to recycling. G recently abandoned this requirement, for not being financially viable.

When separated, car glass has to be sent to specific flat glass recycling plants (different from glass bottle recycling). The use for recycled glass depends on where it was situated in the vehicle. For example, part of it is recycled as glass wool for insulation purposes. Due to extra treatment, windscreens are only suitable to be reused as coloured cathedral glass in Hungary.

In many countries (Fr, UK for instance), car glass is still left in the body car and is landfilled with ASR.

c/ Practices prior to the Directive and main environmental problem linked

Glass was part of the ASR sent to landfill. As inert waste, no environmental problem has been known associated to this practice (except the volume occupied in landfill).

d/ What if no ELV Directive

Glass would still be sent to landfill with ASR as an inert waste.

e/ Benefits to-date due to the Directive

Global environmental impacts and benefits linked to glass recycling

No LCA data are available in the literature to assess if car glass recycling generates more or less impacts than the virgin process avoided.

Other effect

Less inert waste landfilled.

4.13 Topic 9 - Removal of pieces: large plastic components

Key figures

Total plastic pieces:
About 100-140 kg of plastics / ELV
About 15 different resins involved

a/ Requirements of the Directive

In order to promote recycling, Annex 1 of the Directive requires from treatment operations removal of large plastic components (bumpers, dashboard, fluid containers, etc), if these materials are not segregated in the shredding process in such a way that they can be effectively recycled as materials.

b/ Current practices

Table below presents a list of ELV plastic components with main resins used and average weight.

Table 4.2: Plastics by type and application, per car for 2005

PART	MAIN PLASTICS TYPE	WEIGHT IN AVERAGE CAR (kg)
BUMPERS	PP	**10.4**
SEATS	PUR, PP, PA, PVC, ABS	**18.4**
COCKPIT	PP, SMA, ABS, PC, PVC, PUR	**21.3**
FUEL SYSTEMS	PE, POM, PA	**8.6**
BODY (including body panels)	PP, PPE, UP	**10.8**
UNDER THE BONNET COMPONENTS	PA, PP, PBT	**13.8**
INTERIOR TRIM	PP, ABS, POM, PVC, PUR	**31**
ELECTRICAL COMPONENTS	PP, PVC, PA, PBT, PE	**10.3**
EXTERIOR TRIM	ABS, PA, PP, PBT,ASA	**5.1**
LIGHTING	PP,PC, ABS, PMMA, UP	**5.6**
UPHOLSTERY	PUR, PP, PVC	**6.8**
OTHER RESERVOIRS	PP, PE, PA	**1.5**
TOTAL		**143.4**

Source: PlasticsEurope, 2005

There is today a very low recycling rate of plastics from ELV in Europe (0% in most MSs). Dk is the only country where dismantlers are obliged to remove plastics for recycling.

c/ Practices prior to the Directive and main environmental problem linked

Plastics were part of the ASR sent to landfill. Main potential problems are linked to leachate water and pollutant discharge via the leachate water route. A modelling of these impacts is available in the literature and results were used in section 8.1.4.1 and are summarised below (see e/).

Regarding air pollution, no landfill gas -CH_4, CO_2- is usually considered being formed from ELV plastic parts (because no biologically active carbon occurs in these plastics).

Remark: however, in the Fraunhofer study (one of the 2 LCA studies used for the analysis of the 2015 options – see section 8.1), greenhouse gas emissions are attributed to the storage of plastics in landfill over a 100-year period (between 250 and 350 g eq. CO_2 / kg of plastic depending on the resin). To the best of our knowledge, few other studies (if any) allocate greenhouse gas emissions to plastics landfilling.

The other environmental problem known for plastics landfilling is the loss of resources.

d/ What if no ELV Directive

Plastics would still be sent to landfill with ASR.

e/ Potential benefits in case of full compliance with the ELV Directive

Global environmental impacts and benefits

The question is: what are the environmental impacts and benefits of large plastic components recycling compared to their landfilling?

Results presented below are extracted from chapter 8.1 (summarised in 8.1.5) where the environmental impacts and benefits associated with different treatments of plastics are analysed in great detail. This analysis was based on 2 peer-reviewed LCAs (one from Fraunhofer and the other one from Öko-Institut for APME).

The 9 following pieces are concerned:

PP (bumper and air duct)	PUR (seat cushion)
PP/EPDM (bumper)	PA-6.6 GF (hubcap)
PA (intake manifold)	PVC, ABS, PUR, PP-TV (dashboard)
PE (wash fluid tank and lid)	ABS (mirror housing).
PC (headlamp lens)	

8 environmental indicators have been assessed quantitatively:

- Energy consumption (MJ)
- Greenhouse effect (direct, 100 yrs) (g CO_2 eq.)
- Air acidification (g SO_2 eq.)
- Photochemical oxidation (g ethylene eq.)
- Water pollution (critical volume in m^3)
- Eutrophication (g PO4 eq.)
- Municipal waste (kg)

- Hazardous waste (kg)

And 2 environmental impact indicators have been approached qualitatively:

- Non renewable resource depletion
- Land use.

When considering landfill versus mechanical recycling, results can be summarised as follows:

- For all the impact categories and for all the components under study except dashboard, mechanical recycling has a better environmental profile than landfill when considering a substitution rate[47] of 1.
- When the substitution rate is less than 1 (which is more likely in real industrial conditions according to plastic experts), the results are much more contrasted. No general conclusion can be drawn except that the lower the substitution rate, the lower the environmental benefits of mechanical recycling and, under a certain level of substitution rate, benefits can even be replaced by disbenefits which can become higher that landfill impacts (for instance for PUR, this threshold is between 0.65 and 1). This is further analysed through sensitivity analyses about substitution rates in section 8.1.6.
- With respect to dashboard (a mix of resins), this general conclusion has to be moderated as mechanical recycling has a better environmental profile than landfill for all impact categories except for global warming.

As a general conclusion, one can keep in mind that **there are cases where mechanical recycling has a better environmental profile than landfill but this is not always the case** (depends on the resin and the quality of granulates obtained to be recycled, the latest influencing the substitution rate).

Remark: there is one environmental impact category which is never quantified in LCA and which is important when considering landfill: land use. Landfill is known for being detrimental to land use. In cases where recycling is less beneficial than landfill for some impact categories, it is sometimes heard that, still, it is better than landfill for land use impact. In fact, this would need to be demonstrated because all the facilities involved in the recycling system also occupy land.

f/ Benefits to-date due to the Directive

Stakeholders mentioned that the Directive has encouraged the **carrying out of technical feasibility studies about recycling and recovery of large plastic pieces**.

47 The substitution rate relates to the quantity of virgin material (in kg) that can be substituted by 1 kg of recyclates in the end product in order to achieve an equivalent performance. For example, if a 1-kg plastic part made from virgin material could only be substituted by 1 kg of recyclates, then SR=1, whereas if a 500 g plastic part made from virgin material could only be substituted by 1 kg of recyclates, then SR=0.5

4.14 Topic 10 - Removal of pieces: metal components

a/ Requirements of the Directive

In order to promote recycling, Annex 1 of the Directive requires from treatment operations the removal of metal components containing copper, aluminium and magnesium if these metals are not segregated in the shredding process.

b/ Current practices

According to stakeholders interviewed, most of them would be removed already, including in illegal facilities as they have an economical value. But no data are available in the literature to confirm that.

c/ Benefits to-date due to the Directive

No major benefits.

4.15 Summary of environmental benefits to-date

Table 4.3: Summary of environmental impacts and benefits to-date of the ELV Directive

Directive's requirements	**Impacts & Benefits to-date due to the ELV Directive**
100% of ELV are collected and transferred to authorised treatment facilities ATF (Articles 5.1 and 5.2)	A slow but real progressive replacement of illegal treatment facilities by authorised (and thus controlled) ones: **about 50% of ELV are estimated being now treated in ATF in Eu25**
	Even if ATF do not all comply with the directive yet, **the number of ATF adopting the environmental requirements of the Directive has been increasing regularly** (even by anticipation of the Directive in several MSs)
	More and more materials with a 0 or negative value are properly treated and recycled / recovered (not only those with an economical value anymore) (see below an attempt of quantification)
Depollution – Fluids (Annex 1)	Local environmental **benefits** If we anticipate when all the ATF (which receive about 50% of ELV thanks to the directive) comply with the ELV directive, then: About **56 kt of fluids per year** (31 kt of waste oils and 25 kt of water-based fluids containing glycol) **are not released into the environment anymore or left in ASR**, **avoiding the potential risk of contamination of soil, water, and biota** (in particular by heavy metals and PAHs and by ethylene glycol) which can cause adverse effects on **human health** (in particular through food chains for waste oils: a wide range of illnesses, from irritations to cancer, anaemia, skin ulcerations and cardiovascular disease), **acute toxicity and death in aquatic organisms and in fish, poisoning of plants**.
	Global environmental **benefits** If we anticipate when all the ATF (which receive about 50% of ELV thanks to the directive) comply with the ELV directive, then: About **56 kt of fluids are sent to proper management facilities per year** – the energy which can potentially be recovered from the 31 kt of WO would allow **a car to turn around the earth 12,000 times** – from an LCA perspective, WO recovery options exist (regeneration or incineration) which contribute **to avoid global environmental impacts** (such as global warming potential, regional acidifying potential, VOC emission) by comparison with a 'do nothing' situation (where base oil or energy would be produced from raw material instead of from WO) (in other words: recovery options (regeneration or incineration) exist which produce lube oil or energy from WO with less environmental impacts than if this base oil or energy were produced from raw materials)
Depollution – Batteries (Annex 1)	Local environmental **benefits** Assumptions: without the ELV directive, 10-15% of spent batteries would be improperly treated. Thanks to the implementation of the directive, 50% of them have already been captured. – **2 to 4 kt of sulphuric acid per year not released into the environment anymore**, **avoiding the potential risk of contamination of water** directly or from soil – **2 to 4 kt of lead diverted from waste** – **500 to 750,000 millions of batteries recycled in controlled installations** (with no more risk linked to wind dispersal of lead dust, substantial atmospheric emissions (lead-containing dust, soot, SO_2, chlorides, dioxins, etc.), open tipping of residues and waste such as batteries casings).
	Global environmental **impacts** Those generating by batteries recycling, in particular to air.

Directive's requirements	Impacts & Benefits to-date due to the ELV Directive (contd.)
Depollution – Liquified gas tanks (Annex 1)	Local environmental **benefits** **Progressive change in behaviour**, not in the NL where good practices are prior to the directive, but in the other countries where ATF in line with the directive begin to remove LGT (and thus **avoids the risk of explosion during shredding**) and to have **propane neutralised** (rather than being released into the air to create ozone which can cause to skin and eye irritations). However practices are not stabilised yet given that concerned quantities are still low.
Depollution – Air bags (Annex 1)	Local environmental benefits **No major environmental benefits to date** can be attributed to the directive since practices did not evolve significantly yet (proportion of ELV with air bags being still low), **except that solutions are developing on the ground** to remove and neutralise air bags (this will generate more benefits within 5 years or so, when air bags in ELV are more numerous).
Recycling – Catalysts (Annex 1)	Environment benefits None
Recycling – Tyres (Annex 1)	Environment **benefits** The ELV directive probably **contributed**, together with the Landfill directive, **to the disappearance of tyres landfilling and stockpiling practices in ATF in line with the ELV directive**, by establishing clear responsibility upon specific actors (dismantlers and shredders). The role played by the ELV directive in that behavioural change is likely to be different according to MSs (which had different pre-existing practices and legislations). No major impact on the recycling rate yet.
Recycling – Glass (Annex 1)	Following the implementation of the ELV directive, glass recycling is done in some MSs (no statistics found to assess the quantity of glass concerned at the Eu level). In these cases: Global environment impacts and benefits **No LCA data available** in the literature to assess if car glass recycling generates more or less impacts than the virgin process(es) avoided. Other effect (local) **Less inert waste landfilled**. In the other cases, no effect of the directive.
Recycling – Large plastic components (Annex 1)	The Directive has encouraged the **carrying out of technical feasibility studies about recycling and recovery of large plastic pieces**.
Recycling – Metal components (Annex 1)	No major (since were largely recycled yet)

4.16 Environmental impacts & benefits associated with 2006 targets

In this chapter about environmental benefits to date, it was decided to also cover the benefits linked to the recycling and recovery targets set up for 2006 as a related issue.

The methodology we used to analyse this issue is the one we developed to assess possible options for 2015 targets which is presented in section 8.2.2.

To facilitate the reading of the results for the 2006 targets, they are presented in the same part of the report as for the 2015 targets. See section 8.3.

PART C: FUTURE COSTS AND BENEFITS OF THE DIRECTIVE

5 OPTIONS FOR INCREASING RECYCLING AND RECOVERY RATES

5.1 General Approach

This Section examines the current (2005) levels and types of reuse, recycling and recovery of ELVs, based on descriptions of the material composition of a typical ELV. It then describes the possible 'non-Directive' activity, as a baseline from which to assess the required increases in reuse, recycling and recovery to meet the targets in the Directive and hence the need for appropriate technologies and their associated costs.

The section then presents a series of scenarios describing the ways of achieving higher targets for recycling and recovery, by treatment method and by type of material. These scenarios form the basis of the estimated costs in Section 6.0.

5.2 End of Life Vehicle Composition

Table 5.1 indicates the quantities of materials and components in a typical ELV, at the present time and as expected to arise in 2006 and in 2015 (holding the total weight constant) based on previous and current vehicle specifications, taking into account an average vehicle life of 13 years. The figures are the volume of materials per tonne of ELV, and approximate to the quantities in one vehicle given that the average weight of an ELV is approximately one tonne (or one thousand kg).

Table 5.1: Composition of Typical ELV Over Time

	kg per tonne of ELV		
Material / Fraction	**2002**	**2006**	**2015**
Ferrous Metal	680	680	650
Non Ferrous Metal	80	80	90
Plastics and Process Polymers	100	100	120
Tyres	30	30	30
Glass	30	30	30
Batteries	13	13	13
Fluids	17	17	17
Textiles	10	10	10
Rubber	20	20	20
Other	20	20	20
Total	1000	1000	1000

Source: TRL (2003), GHK estimates

Changes in vehicle composition over the past decade has seen the greater use of plastics and non-ferrous metals in place of ferrous metals, so that some change in the composition of a typical ELV is expected between now and 2015. Vehicles are also increasing in size, with the average weight of an ELV, despite the use of lighter materials projected to increase from 951kg in the baseline to 964kg in 2006 and to 1025kg in 2015. Further details are given in Annex 2.

These changes in vehicle composition and weight are driven by a range of factors including safety, fuel efficiency, and consumer preferences. It is reported that part of the reason for the move from ferrous (steel) to non-ferrous (aluminium) metals in construction has been to increase the value of ELVs, thus enabling some financing of the take back provisions and related treatment. The possibility of future vehicle design changes as a result of the Directive cannot be ruled out, but the pressure of other drivers means that it can only be one of several drivers for change.

5.3 Current Practice

Current practice (2005) in the treatment of ELVs varies between EU member states. Table 5.2 gives estimates of current rates of recycling, reuse and recovery, taken from the case studies of selected MS (Annex 4).

Table 5.2: Estimated Current Rates of Recycling, Reuse and Recovery (2005)

Selected Member State	Estimated %
France	75-80
Germany	82
Hungary	75
Netherlands	84
Spain	75
UK	80

Source: Country Case Studies

Three different starting points can be identified for ELV treatment, depending on the MS. These are:

1. *Minimum, market based practice* – c.75% of the ELV is recycled/reused – almost entirely metals but some valuable components such as batteries. This is typical of the starting point in many parts of Southern, Central and Eastern Europe which have been slow to adjust to higher standards.

2. *Market-based practice plus depollution* – c.80% of the ELV is recycled, reused and recovered – in compliance with enhanced legislative requirements - including batteries and fluids, plus tyres as required by the Landfill Directive. This situation is typical of larger MS including the UK, France and Germany.

3. *Advanced standards of ELV treatment* – c.85% of ELV reuse, recycling and recovery. This requires going beyond minimum environmental standards and market-based solutions and recycling some materials at additional cost, in order to meet higher recycling targets. This has been achieved by the Netherlands, largely through the dismantling of ELVs, financed from the use of a charge on new vehicles.

Tables 5.3, 5.4 and 5.5 provide generalised illustrations of these different starting points based on an average ELV of 1,000 kg.

Table 5.3: ELV Treatment under Minimum, "Market-Based" Starting Point (kg)

Material / Fraction	Reuse	Recycling	Recovery	Landfill	Total
Ferrous Metal	31	597	0	22	650
Non Ferrous Metal	9	68	0	14	90
Plastics & Process Polymers	0	0	0	120	120
Tyres	0	0	5	25	30
Glass	0	0	0	30	30
Batteries	1	9	0	3	13
Fluids	5	5	0	7	17
Textiles	0	0	0	10	10
Rubber	0	0	0	20	20
Other	0	0	0	20	20
Total	**46**	**679**	**5**	**270**	**1000**

Option provides for a reuse and recycling rate of 72.5% and a recovery rate of 73%

Table 5.4: ELV Treatment under "Market-Based & Depollution" Starting Point (kg)

Material / Fraction	Reuse	Recycling	Recovery	Landfill	Total
Ferrous Metal	31	612	0	8	650
Non Ferrous Metal	9	79	0	2	90
Plastics & Process Polymers	1	0	0	119	120
Tyres	10	10	10	0	30
Glass	0	0	0	30	30
Batteries	1	12	0	0	13
Fluids	5	12	0	0	17
Textiles	0	0	0	10	10
Rubber	0	0	0	20	20
Other	0	0	0	20	20
Total	**57**	**724**	**10**	**209**	**1000**

Option provides for a reuse and recycling rate of 78.1% and a recovery rate of 79.1%

Table 5.5: ELV Treatment under "Advanced Dismantling" Starting Point (kg)

Material / Fraction	Reuse	Recycling	Recovery	Landfill	Total
Ferrous Metal	31	612	0	8	650
Non Ferrous Metal	9	79	0	2	90
Plastics & Process Polymers	1	11	12	96	120
Tyres	10	10	10	0	30
Glass	1	14	0	15	30
Batteries	1	12	0	0	13
Fluids	5	12	0	0	17
Textiles	0	1	0	9	10
Rubber	0	5	0	15	20
Other	0	0	0	20	20

Total	**58**	**755**	**22**	**165**	**1000**

Option provides for a reuse and recycling rate of 81.3% and a recovery rate of 83.5%

5.4 Review of the Technical Options and Related Costs for Treating ELVs

As described in Section 1.0, ELVs may either be dismantled, to remove valuable parts, or sent directly for shredding, traditionally via a scrap yard. For example, in the UK, it has been estimated that two thirds of ELVs are dismantled while the remaining one third are sent directly to scrap yards from which they are sent for shredding (Defra, 2005).

Depollution may take place either at the dismantling stage or prior to shredding. However, while in the past vehicles were often shredded whole, increased requirements under Annex I of the Directive to remove materials (not just for depollution but also other materials such as tyres and glass for recycling) are likely to encourage increased numbers of vehicles to be dismantled prior to shredding.

ELVs may be received by a large number of organised collection points or small operators, including scrap yards, dismantling businesses, salvage operators and secondary metals businesses. In contrast, shredding plants are large, capital intensive operations and are relatively few in number. Annex 2 provides further details.

The Directive requires ELVs to be treated by authorised treatment facilities (ATFs), and there is a general trend across the EU towards a reduction in the number of dismantling businesses as standards increase and vehicle manufacturers and distributors seek to enter contracts with smaller numbers of approved facilities. A description of the German system is provided to illustrate the intended treatment activity (Table 5.6).

Table 5.6: Steps in German ELV recycling process

Collection	Owner takes ELV to: - collection points set up by producers/importers - independent collection points - directly to a recycling company/dismantler
Dismantling	Task of dismantling company: - drainage of liquids - dismantling of spare parts for resale - collection of recycling materials -collection of tyres for recycling -collection of batteries for recycling -collection of plastic parts for re-use or recycling - pressing of car wrecks for delivery to shredders
Shredder company	Task of shredding companies: - shredding - magnetic separation of ferrous metals - separation and sorting of non-ferrous metals - separation and sorting of shredder waste - collection of high energy content residue for recovery

Source: BVSE Altautoverwertung

Annexes 2 and 3 provide a more detailed analysis of the various technical options to increase rates of recycling and recovery and their related costs. We summarise the main options below.

5.4.1 Depollution

Article 6 of the Directive requires ELVs to be depolluted, and relates mainly to the collection and treatment of batteries and various fluids. Annex 2 of the Directive sets out detailed treatment requirements for depollution. The technical response has been the design of 'depollution rigs' which enable the various fluids to be collected separately. The costs of depollution have been estimated at 30 euro per ELV in France (Ademe, 2003). The Stakeholder Group report puts the average cost of depollution and essential dismantling at 40-80 euro per ELV, including administration costs.

Since depollution is a legal requirement to be implemented as soon as possible, we have assumed that this treatment activity and related costs are part of the baseline from which the additional costs of meeting the recycling and recovery targets (in 2006 and 2015) are to be estimated.

5.4.2 Dismantling

Dismantling involves the removal of the most valuable or required parts of the vehicle for reuse or reprocessing including engines, gearboxes, radiators, carburettors, alternators, starter motors, distributors and headlamps. If not reused, larger metal parts such as engines, gearboxes and carburettors are often removed and sent to specialist reprocessors for metal recovery.

The Netherlands has made more progress than any other Member State in promoting reuse and recycling of ELVs through dismantling, which has been central to the achievement of a current 84% reuse/recycling rate. A variety of components including batteries, tyres, inner tubes, fuel, bumpers, glass, grilles, coolants, coconut fibre, refrigerants, LPG tanks, oil, oil filters, PUR foam, braking fluid, rubber strips, windscreen washer fluid, safety belts and hubcaps are routinely removed and recycled. Standard quantities per ELV are set for the above materials and components, and payments are claimed on this basis, under commission from ARN. More valuable, generally metal parts may be sold in the market.

Costs of dismantling depend on car type and depth of dismantling. In Germany this cost has been estimated at €250-350 per ELV.[48] In France, ADEME estimated treatment costs for dismantlers (including staff expenses, overheads and depreciation) averaged €330 per ELV.

Dismantling costs are expected to be covered (depending on the depth of dismantling) by revenues received from the sale of parts and materials. In France the value of parts sold averaged €495 per vehicle, with a further €23 received for the sale of the body shell.

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[48] BDSV (2002)“ *Nachteil für den Mittelstand? Entsorgung von Altautos neugeordnet*“, press release 25 Januaru 2002

Evidence from MS suggests that the costs of recycling material from ELVs through dismantling is highly variable, reflecting the steep marginal cost curve for dismantling operations. Some larger parts (e.g. plastic bumpers and larger sections of glass) may be removed relatively cost effectively, while marginal costs rise steeply as more material is removed. The figures suggest that small quantities (maximum 30-40kg) of more easily removed materials may be dismantled at a moderate cost of 0.2 to 0.3 euro per kg, while removal of larger quantities is likely to raise marginal costs to more than 1 euro per kg.

Reference to published estimates of the time taken for increasing levels of dismantling of plastics for recycling indicates the rapid increase in effort beyond a limited point of dismantling.[49] Using the average EU hourly labour cost[50] this effort approximates to around 1 euro per kg after the dismantling of approximately 70kg (Figure 5.2).

Figure 5.1: Dismantling time for a car (total plastics 160kg)

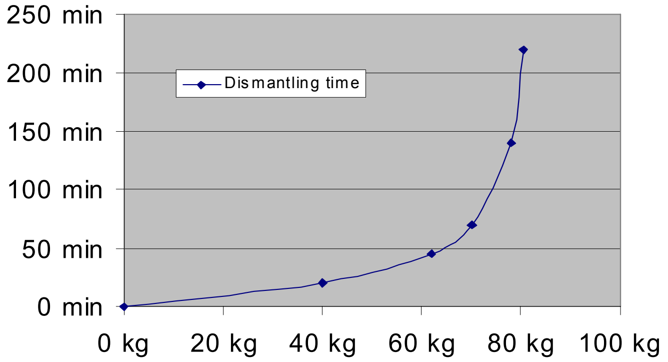
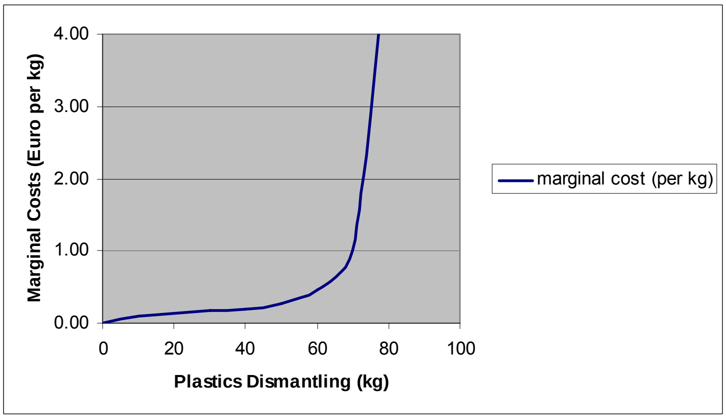


Figure 5.2: Marginal Costs of Plastics Dismantling



[49] Source: PlasticsEurope Eco-efficiency considerations on some prevention and recycling scenarios. September 2004

[50] Average EU25 labour cost per hour for an industrial worker is estimated at 20.22 euro (including direct and indirect costs), 2003. Source: Eurostat, annual labour cost data

5.4.3 *Shredding, Recycling and Recovery*

The shells of all ELVs treated in the EU are eventually shredded, after depollution and usually after removal of valuable parts through dismantling. This is a capital intensive process undertaken by a limited number of plants in each Member State.

ELVs are shredded (often in combination with other sources of metals) and the resulting fragments sorted into ferrous metal, non-ferrous metal and shredder residue. A shredder is able to recover the majority of the metal content (ferrous and non-ferrous) of a vehicle by its magnetic and density properties. The non-metallic fraction (Auto Shredder Residue - ASR) comprises materials such as plastic, foam, glass, rubber and textiles. ASR accounts for between 15% and 25% of the weight of an ELV, depending on the proportion of materials recovered. Though plants are being developed that can process ASR for recycling and/or for energy recovery, the majority is currently disposed in landfill sites.

The challenge of meeting the higher recycling and recovery targets will require increased treatment of ASR. The technologies for doing so (collectively described as post shredder technologies (PST)) are still being developed. It is therefore difficult to estimate how effective and costly these options will be. To address this uncertainty the work has undertaken a detailed review of these emerging technologies. This review is presented in Annex 3. Table 5.7 presents an overview of these post shredder technologies.

In summary there are two main categories of technology, those based on mechanical sorting of the waste into different fractions that can be recycled and sold; and those based on thermal treatment of the waste stream to generate feedstocks for energy generation. In the case of mechanical separation, the main products are plastic granulates (for recycling in plastic products), shredder fibres (for use in sewage treatment) and sand (for use in construction). The technology still generates wastes that require disposal in incinerators or landfill sites. Thermal treatment generates feedstocks for energy production (eg gas) or energy directly from heat generation. They also produce inert materials for use in construction or for disposal.

The technical review of PSTs, based on the available information from operators and technology owners allows some appreciation of the environmental effectiveness of the PSTs. The information suggests that PSTs range in their reported effectiveness in terms of the overall rates of recycling and recovery of material treated, from around 50% (Galloo and Citron – although the Citron process is intended to recover the additional 50% waste material when operating at industrial scale) to 100% (Sult and R-Plus).

In terms of recycling, the reported effectiveness of mechanical separation technologies ranges from 74% (Sicon) to 100% (R-Plus). The thermal treatment processes are also intended to recycle some material, principally the remaining metallic residues. These PSTs achieve recycling rates of between 8% (Schwarze-Pumpe) and 39% (Galloo). The planned Citron plant is intended to achieve a recycling rate of 50%.

The PSTs are designed to operate after depollution, commercial dismantling and shredding. Thus the PSTs are designed to deal with the remaining 20% by weight of the average ELV. The rates of recycling and recovery of the PSTs are summarised in Table 5.8, based on the treatment of the residual 20%.

Table 5.7: Overview of Post Shredder Technologies as Currently Developed and Tested

Name of Technology / Developer	Type of Technology	Level of Technology Development	Approximate Outputs from Process	Indicative Gate Fee (euro per tonne of ASR)
Citron	Thermal treatment – oxyreducer	1 Plant (130,000 tonne, 12,000 ASR). Plans for a 500,000 tonne (120,000 ASR) plant.	Current – Ca Fe concentrate 45%, zinc concentrate 4.3%, mercury 0.7%, wastes 50%. Plan – Ca Fe concentrate 45%, Zinc concentrate 4.3%, mercury 0.7%, recovery 50%	100 – 200 (excluding energy sales)
VW - Sicon	Mechanical separation	1 plant (8,000 tonne) plus 2 under construction. Plans for a 100,000 tonne plant	Shredder granules 36%, shredder fibres 31%, metals 8%, wastes 26%	20 – 50
Galloo	Mechanical Separation	Operating plants	Recycled plastics 9%, metals 30%, refuse derived fuel 13%, wastes 48%	Not available
Sult	Mechanical separation	Operating plant in Japan	Organic (plastic) 50%, minerals 20%, metals 10%, water 20%	100
R-Plus	Mechanical separation	Operating plants	Organic fraction 60%, minerals 35%, metals 5%	90
TwinRec	Thermal treatment – gasifier	Operating plants in Japan	Metals 8%, glass granulate 25%, recovery 52%, wastes 15%	120 – 200
SVZ Schwarze Pumpe	Thermal treatment – gasifier	Industrial trial plant	Synthetic gas 75%, metals 8%, wastes 17%	Not available
Reshment	Mechanical separation & thermal treatment	No pilot or trial plants	Not available	75 – 140

Note: Gate fee is the charge to waste producers for treatment of the waste stream. The fee is determined by the treatment costs less income from sales of materials or energy. Transport costs are borne by the waste producer.

Table 5.8: Recycling and Recovery Rates of ELVs Using PSTs with Current Market and Depollution Practices

Technology / Developer	Type of Technology	Overall Recycling & Recovery Rate (%)	Recycling Rate (%)
VW – Sicon	Mechanical separation	95%	95%
Galloo	Mechanical separation	90%	88%
Sult	Mechanical separation	100%	96%
R-Plus	Mechanical separation	100%	100%
Citron – planned	Thermal treatment – oxyreducer	100%	90%
TwinRec	Thermal treatment - gasifier	97%	87%
SVZ Schwarze Pumpe	Thermal treatment - gasifier	97%	82%

Source: Annex 3: PST Technical Review

This shows that all the technologies (with the exception of Galloo), based on the information provided, are able (with market and depollution practices) to achieve overall rates of recycling and recovery of 95% or more. It also indicates that all the PSTs (with the exception of Schwarze-Pumpe) are able to achieve in excess of an 85% recycling rate. In the case of thermal treatment plants this is mainly because of the separation and recycling of residual metal fractions. In the case of mechanical separation plants the overall rates are achieved through recycling of all fractions, especially plastics.

The technical review indicates that mechanical separation processes are estimated at the present time to be cheaper than thermal treatment processes. The approximate costs per tonne ASR for mechanical separation range from as low as 20 euro (Sicon) to 100 euro (Sult / R-Plus). The approximate costs for thermal treatment range from 75 euro to 200 euro (Citron and Twin Rec). These costs are used in the cost assessment in Section 6.

The costs are sensitive to economies of scale, with higher costs associated with smaller plants. The costs are also sensitive to assumed levels and types of recycled materials and energy recovered and associated prices.

The evidence from reviews of other environmental technologies that have emerged in response to higher environmental standards is that initial cost estimates, partly based on pilot and trial operations, tend to overstate costs and understate the effectiveness of technologies when operated at industrial scale with tested techniques. Work undertaken to review costs of selected technologies over time suggest that costs decline by between 4% and 10% per year[51]. In the assessment of costs we have assumed a conservative reduction of 10% in total from the costs to achieve 2006 targets by 2015, but tested the sensitivity of resulting cost estimates to a reduction of 50%.

The emergence of the PSTs suggest the need for a review of Annex 1 (4) of the ELV Directive. This requires certain treatment operations including the removal of glass, tyres and large plastic components if these materials are not segregated in the PST and effectively recycled. The Annex therefore introduces certain restrictions on the treatment operations, with an implied preference for dismantling, when greater flexibility could allow a more effective technology to emerge, with no loss of environmental benefit. Work to examine the relative environmental benefits of treatment options, and especially of energy recovery compared to recycling, is presented in Section 7.0, below.

Without any targets for recycling and recovery, given that there are no industrial scale plants to demonstrate and 'prove' the technology, there is the risk that the PSTs reviewed would not survive as commercially attractive options. At the present time only the Sicon process would be able to compete with landfill operators. However, in the context of the Directive, the only alternative way of attempting to secure compliance is through increased dismantling rates. The evidence from the case studies and other reported material is that dismantling costs for the rates of material recycling required for 2015 would be well in excess of the costs currently estimated for PSTs. Maintenance of the targets in the Directive would therefore be technology forcing and provide a strong legislative basis for continued investment.

There are also a number of factors that are likely to sustain continued investment in PSTs

- The requirement for free take back of ELVs creates a strong economic incentive for vehicle producers to identify the least cost routes for meeting the targets, and for supporting through contractual obligations investment in treatment options.
- The provisions of the Landfill Directive and limited landfill and waste incineration capacity necessitating new investment in capacity to new standards will raise the costs of disposal in real terms over the next ten years. This will improve the competitiveness of PSTs ahead of the introduction of statutory targets in 2015.
- The development and subsequent licensing of PSTs will require significant capital investment. Companies already operating shredding processes are the businesses logistically best placed to invest in PSTs. Shredding processes are themselves significant capital intensive operations and require businesses that are able to attract and manage investment in technology.
- The existence of strong regulations on ASR disposal in non-EU countries (especially Switzerland and Japan) which has already forced PST investment

[51] TME and RIVM technology review. Annex 1: Impact Assessment of Environmental Policy on Business – Methodological Issues

and which will continue to provide a market for EU technology developed and/or lead to technology development capable of transfer to the EU.

5.4.4 Disposal to Landfill

The level of reuse and recycling of ELVs has always been high, because of the market price for scrap metal, with approximately 75% by weight of ELVs reused and recycled by the market. Depollution measures increase this to approximately 80% by weight. There is therefore approximately 20% or 200kg of material from an ELV which is disposed of to landfill, in the absence of further incentives or regulation. This material is largely in the form of the ASR resulting from the shredding process and the separation of scrap metal. Landfill disposal costs for ASR range from 30 euro per tonne in lower cost MS (mainly central and eastern Europe) to around 60 euro per tonne in medium cost MS (eg France UK, Italy) and over 100 euro per tonne in high cost MS (eg Germany, Denmark, Sweden).

As well as a concern to reduce the waste of resources associated with this disposal, there has also been a concern with the environmental effects of disposing of this waste in landfill sites resulting from the cross contamination of the material in the shredding process, as a result of contact with pollutants from other waste streams such as PCBs. This has led some MS to consider a ban on the disposal of ASR to MSW landfill sites, and requiring further treatment (incineration) and / or disposal as hazardous waste. We have discussed this issue with a number of landfill operators who consider that the contamination is generally insufficient (less than 50 ppm) to warrant classification of ASR mixed with non-ASR as hazardous waste.

The costs of landfill disposal for ASR have therefore been rising in those MS with a concern over the environmental impact of ASR, and also because of an interest in providing an incentive for alternative treatment technologies. Existing variations between MS in the landfill costs of ASR may therefore widen still further. These variations already give rise to some trade in body shells with exports to lower cost MS, for example from France to Spain. Further trade can therefore be expected until there is some convergence in disposal cost.

5.4.5 Disposal to Incinerator

At the present time there is only very limited disposal of ASR by incineration. There have been trials in Germany to examine the incineration of shredder residue (SR) with other domestic waste, including up to 30% of SR in the mix. Charges for waste incineration in Germany are €70-300/tonne. For new plants costs of €100/tonne are regarded as realistic. However, due to the change to the landfill regulation from June 2005, it is expected that there is insufficient capacity within Germany for the necessary treatment of domestic waste. This will place significant constraints on the capacity to incinerate ASR, increasing interest in the development of dedicated ASR treatment plant. Similar constraints on capacity also exist in the UK.

The lack of public support for major expansions of waste incineration capacity (eg in the UK and Hungary) suggests that the thermal treatment of SR will need to be developed and marketed as a specific technology distinct from municipal waste incinerators, promoting the plants as power or recovery plants to avoid the stigma attached to waste incineration.

The interest in the use of ASR for energy recovery, given the relatively high calorific value of ASR, resulting mainly from the plastic fraction, is reflected in the emerging PSTs reviewed above, section (5.4.3). Treatment costs of around 100 to 200 euro per tonne (net of income from the sale of energy) are indicated from the PST review.

5.5 Future Recycling Markets

The review of technical options in the previous section indicates plausible treatment routes for meeting the targets set out in the Directive, recognising the largely untested nature of post shredder technologies (PSTs) as industrial scale operations. The costs of these PSTs as charged to waste producers will depend on the gate fee, which is determined by the processing costs, less the income received from sale of recyclates or feedstocks.

In terms of the potential income to PSTs, the major risk seems to lie with plastics recyclates, as the major fraction in terms of both weight and income. Some of the mechanical treatment processes produce plastic fractions (PVC, PE, PP and EPDM) which are sold on the plastic recyclates market.

Around 58,000 tonnes of automotive plastic was estimated to have been mechanically recycled in 2002. This represents 0.15% of European plastics consumption and 2% of the current recyclates market. Plastics make up approximately 100kg per ELV (Table 5.1). This is projected to rise to around 120kg per ELV by 2015. Assuming 10m ELVs are treated per year, and all this material were to be collected, this would give rise to some 1 million tonnes, adding around 30% to current EU volumes of recycled plastics of 3.1 million tones, which would otherwise be disposed of to landfill.

Discussions with the potential PST operators indicate that they are confident that there will continue to be a market for these plastic recyclates. This is supported by the current growth of the virgin and recycled plastics market, as presented below.

5.5.1 Projected recovered plastic demand

The world production of plastics has increased over five fold since 1970 and 169m tonnes were produced in 2002[52]. Consumption of plastics is also growing continually and in Europe rose 5.6% between 2001 and 2003 to 40m tonnes, representing around a quarter of global demand, despite difficult economic conditions.

In 2003, around 3.1 million tonnes of plastic from European waste streams were mechanically recycled: 0.4m tonnes were recycled outside Europe where there is a strong demand for secondary plastics raw material; 2.7 million tonnes were mechanically recycled in Europe into recycled granules or products.

A significant amount of the EU's recovered plastic is being exported for reprocessing in Asia, particularly China, Hong Kong and India, where there is growing demand for recycled plastic to make consumer goods, and which have invested in the necessary reprocessing capacity. In 2004, around 1.36m tonnes of plastic worth around €310million was exported from Europe to these three destinations[53] (Table 5.9). This

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[52] PlasticsEurope, An analysis of plastics consumption and recovery in Europe, Summer 2004

[53] Source: EU Eurostat online, http://epp.eurostat.cec.eu.int/

demand has been growing over the last five years and demand from these countries is expected to continue to rise over next ten to fifteen years.[54]. Table 5.8 shows the main destination of recycled plastics exported from the EU.

Table 5.9: EU Exports of Recycled Plastics

1999			2004		
Importing Country	Trade Value (£)	Net Weight (tonnes)	Importing Country	Trade Value (£)	Net Weight (tonnes)
TOTAL	171,237,461	1,029,936	TOTAL	438,970,977	2,363,073
Hong Kong	33,512,693	347,290	Hong Kong	111,347,423	719,586
US	23,841,384	71,673	China	97,263,865	526,201
India	3,705,026	30,691	US	17,298,537	70,408
China	2,366,815	18,264	India	12,406,055	88,890
Others	110,178,357	580,282	Others	200,655,095	957,989

Source: EU Eurostat online, http://epp.eurostat.cec.eu.int/

The study findings also suggest that there is growing demand within Europe for recovered plastics, especially semi-processed (granulated) products of the type produced from the PST process.

5.5.2 Oil prices

The capacity of markets to absorb the increase in recyclates (taking into account the global market demand and the scope for exports), depends in part upon the costs of virgin plastic, which in turn depends upon the future cost of oil.

Virgin plastic resins are produced in large-capacity facilities from the monomers that are the building blocks to plastic polymers. Oil and gas are the main raw materials for producing the monomers. The price of oil is therefore very closely linked to the price of plastics, which in turn sets the price for recycled plastics, which are generally cheaper. Although it is difficult to predict with any confidence the price of oil after 2010, the general consensus [55] is that:

4. Prices will stay high i.e. above $40/barrel. Even though stocks are adequate, the market will always be nervous about the lack of spare capacity/geopolitical risks/over reliance on Saudi Arabia and this will provide a floor to prices.

5. Demand from USA / China / other developing nations is going to continue to grow for the forseeable future. The question is whether supply will grow at a faster, equivalent or slower pace. Crude supply is currently adequate but there is not enough refining capacity which is why product prices (gasoline, heating oil) have reached extremely high levels. However, with refining margins high, investment is

[54] Assessment of the export market for recycled plastics, for WRAP, final report due May 2006, GHK. Exports have increased by 57% in quantity and 61% in value since 1999.

[55] As reported through discussion with oil futures trader, personal communication

being made in the refining system and new refineries should come online around 2009/2010. In the meantime price spikes are likely to occur.

6. The real worry is that there is only a finite supply of crude oil. There are a wide range of estimates as to when 'peak oil' will arrive but realistically it will probably be around 2012-2015. At this point it will be impossible to say how high prices could go.

7. The risks are therefore heavily skewed towards higher oil prices and it could be that prices of $100/barrel oil might be reached before the end of the decade.

Virgin plastic prices are also increasing, reflecting both higher costs but also increasing demand, with commodities seen as the new 'investment of choice' for hedge funds, which is helping to keep prices high.

The general trend towards higher oil and plastics prices suggests that there is room for a growing plastic recyclates market over the next fifteen years.

5.6 Treatment Options as the Basis of the Cost Assessment

The calculation of the costs (or cost savings) of achieving higher targets of recycling and recovery require specific treatment options to be specified. This specification has to be made in recognition that the selection of treatment methods to meet the 2006 targets is either yet to be determined by MS or is changing from the use of present dismantling practices (eg Netherlands and Germany).

This in turn means that there is the possibility of some mix of treatment options, with some investment to meet the 2006 targets, followed by further investment to meet higher targets. If MS require the waste treatment sector to employ this strategy this is likely to involve some additional dismantling followed by investment in PSTs. Given the different forms of PST it may also be possible that there could be a mix of PSTs, although the benefits of economies of scale mean that fewer larger plants will be preferred reducing the scope to have a mix of different PSTs.

Given the ability of PSTs (as so far evidenced in the technical review and summarised in Table 5.8 above) to achieve high rates of recycling and / or recovery and the high relative costs of dismantling it seems more likely that MS will allow the waste treatment sector to defer major investment in options to secure the 2006 targets until the technical and economic feasibility of PSTs has been demonstrated and then to chose PST options consistent with EU and MS objectives. This strategy will mean some delay in achieving 2006 targets, but may also mean that the option of achieving higher targets is available before 2015.

Based on the review of options in Section 5.4 above there are three main technical options available to meet higher targets and which form the basis of the cost assessment:

- Dismantling for reuse and recycling
- Mechanical separation and recycling of ASR
- Thermal treatment for energy recovery (with some separation and recycling) of ASR

This cost assessment is presented in the next section.

5.7 Scenarios Describing the Nature of Treatment to Achieve Higher Recycling and Recovery Rates

As the basis for providing an estimate of the economic and environmental impacts of increased recycling and recovery, we have prepared a number of scenarios describing the nature of treatment, by material. Each scenario describes a different treatment route for an ELV consistent with either 2006 targets (Table 5.10) or 2015 targets (Table 5.11). These scenarios allow estimates of the change in treatment for each material type with higher recycling and recovery rates and provides the basis for the calculation of the associated environmental impacts of higher targets and the illustrative economic impacts.

The scenarios presented in Tables 5.10 and Tables 5.11 show the percentage of each material which is reused, recycled, recovered and landfilled. Based on the weight and composition of the ELV this allows, in the bottom half of the table, an estimate of the volume of each material (in kilograms) which is reused, recycled, recovered and landfilled.

The scenarios in Table 5.10 describe alternative means of achieving 2006 targets; they are not descriptions of current activity (which is summarised in Section 5.3). The scenarios in Table 5.11 describe alternative means of achieving higher targets using either PSTs based on mechanical separation or thermal treatment. A variant (mixed option) of the thermal treatment route with an increased recycling rate has also been examined. The breakdown of treatment by material type is based on the review of current practice, and the information given in the technical review on the treatment of different materials by PSTs.

These scenarios should be seen as indicative descriptions rather than precise specifications of the outcomes of treatment routes and are employed for the purposes of estimating the impacts of higher recycling and recovery rates.

Tables 5.12 and 5.13 indicate the changes in treatment taking place between current practice and 2006; and between 2006 and 2015, respectively.

The change to 2006 target rates from current practice has been examined in three scenarios (as described in Table 5.10, scenarios 1,2,3) each describing one of three treatment routes: increased dismantling, use of PST employing mechanical separation; and PST employing thermal treatment. The change from 2006 to 2015 is based on two treatment routes: PST employing mechanical separation (Scenario A); and PST employing thermal treatment (Scenario B) (as described in Table 5.11).

The route chosen to reach 2006 targets influences the treatment pattern in 2015. For example use of dismantling to reach 2006 targets followed by the use of PST based on mechanical separation will yield a different outcome compared to if mechanical separation is used to meet both 2006 targets and higher targets.

As a consequence there are in theory six (3x2) routes to achieving higher rates (labelled Scenarios 1A, 2A, 3A; 1B,2B, 3B). However, dismantling has been shown to be the most expensive option and the two routes based on dismantling are unlikely to occur. The route which starts with thermal treatment and then uses mechanical separation might also be considered unlikely, given the need to increase recycling rates as well as overall recycling and recovery rates. The more likely routes are the three remaining routes highlighted in Table 5.12 (Scenarios 2A, 2B, 3B).

The main results from this analysis are:

- Some residual metals are landfilled in 2006, but all metals are fully recycled by 2015 – note also that due to changes in weight and composition there is an increase in the volume of metals available for recycling
- The main contribution to meeting higher targets is the increased recycling and recovery of plastics. This is the case whether mechanical separation or thermal treatment is used, but is slightly greater in the case of mechanical separation.
- Treatment routes using mechanical separation are assumed to treat a lower share of glass, and to recycle rather than recover tyres.
- Other fractions (eg other rubber, textiles) are not present in large enough volumes to make a significant difference to the achievement of the targets.

Table 5.10: ELV Treatment Scenarios to Meet 2006 Recycling and Recovery Targets

	Scenario 1: Dismantling Route				Scenario 2: PST – Mechanical Separation				Scenario 3: PST – Thermal Treatment			
Material / Fraction	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill
Ferrous Metal	5%	94%	0%	1%	5%	94%	0%	1%	5%	94%	0%	1%
Non Ferrous Metal	10%	88%	0%	3%	10%	89%	0%	1%	10%	90%	0%	0%
Plastics & Process Polymers	1%	9%	10%	81%	1%	27%	0%	72%	1%	0%	21%	79%
Tyres	33%	33%	33%	0%	33%	33%	33%	0%	33%	33%	33%	0%
Glass	2%	40%	0%	59%	2%	0%	0%	99%	2%	16%	0%	83%
Batteries	8%	92%	0%	0%	8%	92%	0%	0%	8%	92%	0%	0%
Fluids	29%	71%	0%	0%	29%	71%	0%	0%	29%	71%	0%	0%
Textiles	0%	10%	0%	90%	0%	30%	0%	70%	0%	0%	20%	80%
Rubber	0%	25%	0%	75%	0%	25%	0%	75%	0%	0%	20%	80%
Other	0%	0%	0%	100%	0%	0%	0%	100%	0%	16%	0%	84%
Kg	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill
Ferrous Metal	31	619	0	6	31	619	0	6	31	619	0	6
Non Ferrous Metal	8	67	0	2	8	69	0	0	8	69	0	0
Plastics & Process Polymers	0	9	10	78	0	26	0	70	0	0	20	76
Tyres	10	10	10	0	10	10	10	0	10	10	10	0
Glass	0	12	0	17	0	0	0	28	0	5	0	24
Batteries	1	12	0	0	1	12	0	0	1	12	0	0
Fluids	5	12	0	0	5	12	0	0	5	12	0	0
Textiles	0	1	0	9	0	3	0	7	0	0	2	8
Rubber	0	5	0	14	0	5	0	14	0	0	4	15
Other	0	0	0	19	0	0	0	19	0	3	0	16
Total	55	745	19	145	55	755	10	145	55	729	36	145
Reuse & Recycling	RR	83.0%			RR	84.0%			RR	81.3%		
Reuse & Recycling & Recovery	RRR	85.0%			RRR	85.0%			RRR	85.0%		

Figures may not sum due to rounding. Assumes an average ELV weight of 964 kg

Table 5.11: ELV Treatment Scenarios Describing Means of Achieving Higher Reuse, Recycling and Recovery Rates

	Scenario A: PST – Mechanical Separation				Scenario B: PST – Thermal Treatment				Scenario C: PST – Mixed Options*			
Material / Fraction	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill
Ferrous Metal	5%	95%	0%	0%	5%	95%	0%	0%	5%	95%	0%	0%
Non Ferrous Metal	10%	90%	0%	0%	10%	90%	0%	0%	10%	90%	0%	0%
Plastics & Process Polymers	1%	100%	0%	0%	1%	14%	60%	26%	1%	66%	34%	0%
Tyres	33%	67%	0%	0%	33%	33%	33%	0%	33%	33%	33%	0%
Glass	2%	0%	0%	99%	2%	76%	0%	23%	2%	0%	0%	99%
Batteries	8%	92%	0%	0%	8%	92%	0%	0%	8%	92%	0%	0%
Fluids	29%	71%	0%	0%	29%	71%	0%	0%	29%	71%	0%	0%
Textiles	0%	98%	0%	2%	0%	14%	60%	26%	0%	98%	0%	2%
Rubber	0%	98%	0%	2%	0%	14%	60%	26%	0%	98%	0%	2%
Other	0%	1%	0%	99%	0%	76%	0%	24%	0%	1%	0%	99%
Kg	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill
Ferrous Metal	31	635	0	0	31	635	0	0	31	635	0	0
Non Ferrous Metal	9	83	0	0	9	83	0	0	9	83	0	0
Plastics & Process Polymers	1	122	0	0	1	17	74	31	1	81	41	0
Tyres	10	21	0	0	10	10	10	0	10	10	10	0
Glass	0	0	0	30	0	23	0	7	0	0	0	30
Batteries	1	12	0	0	1	12	0	0	1	12	0	0
Fluids	5	12	0	0	5	12	0	0	5	12	0	0
Textiles	0	10	0	0	0	1	6	3	0	10	0	0
Rubber	0	20	0	0	0	3	12	5	0	20	0	0
Other	0	0	0	20	0	16	0	5	0	0	0	20
Total	58	916	0	51	58	813	103	51	58	864	51	51
Reuse & Recycling	RR	95%			RR	85%			RR	90%		
Reuse & Recycling & Recovery	RRR	95%			RRR	95%			RRR	95%		

* Mixed option – is a scenario based on the thermal treatment option but assumes that a greater share of plastics is able to be separated and recycled . Figures may not sum due to rounding. Assumes an average ELV weight of 1025 kg

Table 5.12: Changes in Materials Treated, 2006 Compared to Current 'Market & Depollution' Practice

	Dismantling					PST – Mechanical Separation					PST – Thermal Treatment				
	Reuse	Recycling	Recovery	Landfill	Total	Reuse	Recycling	Recovery	Landfill	Total	Reuse	Recycling	Recovery	Landfill	Total
Ferrous Metal	0	10	0	-2	9	0	10	0	-2	9	0	10	0	-2	9
Non Ferrous	0	1	0	0	1	0	2	0	-1	1	0	3	0	-2	1
Plastics	0	9	10	-17	1	0	26	0	-25	1	0	0	20	-19	1
Tyres	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glass	0	12	0	-11	0	0	0	0	0	0	0	5	0	-4	0
Batteries	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fluids	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Textiles	0	1	0	-1	0	0	3	0	-3	0	0	0	2	-2	0
Rubber	0	5	0	-5	0	0	5	0	-5	0	0	0	4	-4	0
Other	0	0	0	0	0	0	0	0	0	0	0	3	0	-3	0
Total	1	38	10	-35	13	1	47	0	-35	13	1	21	26	-35	13

Figures may not sum due to rounding

Note that due to change in weight 13 kg of additional material, mainly metals, requires treatment

Table 5.13: Changes in Materials Treated at Higher Targets Compared to 2006

Based on PST – Mechanical Separation (Scenario A)

	Dismantling (Scenario 1A)	**PST – Mechanical Separation (Scenario 2A)**	PST – Thermal Treatment (scenario (3A)

Fraction	Reuse	Recycle	Recover	Landfill	Total	**Reuse**	**Recycle**	**Recover**	**Landfill**	**Total**	Reuse	Recycle	Recover	Landfill	Total
Ferrous Metal	1	16	0	-6	11	**1**	**16**	**0**	**-6**	**11**	1	16	0	-6	11
Non Ferrous	2	16	0	-2	15	**2**	**14**	**0**	**0**	**15**	2	14	0	0	15
Plastics	0	114	-10	-78	27	**0**	**96**	**0**	**-70**	**27**	0	122	-20	-76	27
Tyres	1	11	-10	0	2	**1**	**11**	**-10**	**0**	**2**	1	11	-10	0	2
Glass	0	-12	0	13	2	**0**	**0**	**0**	**2**	**2**	0	-5	0	6	2
Batteries	0	1	0	0	1	**0**	**1**	**0**	**0**	**1**	0	1	0	0	1
Fluids	0	1	0	0	1	**0**	**1**	**0**	**0**	**1**	0	1	0	0	1
Textiles	0	9	0	-8	1	**0**	**7**	**0**	**-7**	**1**	0	10	-2	-8	1
Rubber	0	15	0	-14	1	**0**	**15**	**0**	**-14**	**1**	0	20	-4	-15	1
Other	0	0	0	1	1	**0**	**0**	**0**	**1**	**1**	0	-3	0	4	1
Total	3	171	-19	-94	61	**3**	**161**	**-10**	**-94**	**61**	3	187	-36	-94	61

Based on PST – Thermal Treatment (Scenario B)

	Dismantling (Scenario 1B)					**PST – Mechanical Separation (Scenario 2B)**					**PST – Thermal Treatment (Scenario 3B)**				
Fraction	Reuse	Recycle	Recover	Landfill	Total	**Reuse**	**Recycle**	**Recover**	**Landfill**	**Total**	**Reuse**	**Recycle**	**Recover**	**Landfill**	**Total**
Ferrous Metal	1	16	0	-6	11	**1**	**16**	**0**	**-6**	**11**	**1**	**16**	**0**	**-6**	**11**
Non Ferrous	2	16	0	-2	15	**2**	**14**	**0**	**0**	**15**	**2**	**14**	**0**	**0**	**15**
Plastics	0	9	64	-46	27	**0**	**-9**	**74**	**-38**	**27**	**0**	**17**	**54**	**-44**	**27**
Tyres	1	1	1	0	2	**1**	**1**	**1**	**0**	**2**	**1**	**1**	**1**	**0**	**2**
Glass	0	12	0	-10	2	**0**	**23**	**0**	**-22**	**2**	**0**	**19**	**0**	**-17**	**2**
Batteries	0	1	0	0	1	**0**	**1**	**0**	**0**	**1**	**0**	**1**	**0**	**0**	**1**
Fluids	0	1	0	0	1	**0**	**1**	**0**	**0**	**1**	**0**	**1**	**0**	**0**	**1**
Textiles	0	0	6	-6	1	**0**	**-1**	**6**	**-4**	**1**	**0**	**1**	**4**	**-5**	**1**
Rubber	0	-2	12	-9	1	**0**	**-2**	**12**	**-9**	**1**	**0**	**3**	**8**	**-10**	**1**
Other	0	16	0	-14	1	**0**	**16**	**0**	**-14**	**1**	**0**	**12**	**0**	**-11**	**1**
Total	3	68	83	-94	61	**3**	**59**	**93**	**-94**	**61**	**3**	**85**	**67**	**-94**	**61**

Figures may not sum due to rounding. Change in vehicle weight adds 61kg by 2015 from 2006

6 ESTIMATION OF THE COSTS OF INCREASED RATES OF RECYCLING AND RECOVERY

6.1 General Approach

The approach to the calculation of the costs of meeting higher targets is based on first estimating the types of activity to comply with the 2006 targets and associated costs; and then estimating the activity required and associated costs to attain higher rates. The difference between the two sets of costs indicates the costs of meeting the 2015 targets compared to the 2006 targets.

Note that the approach assumes that additional treatment only takes place as a result of the need to meet higher targets, and that the related costs are all attributable to these targets. To the extent that the treatment capacity is commissioned to meet the 2006 targets and is capable of achieving higher rates of recycling / recovery than required in 2006 there is an argument that there are no additional costs attributable to the higher targets. An intermediate position is that the fixed costs are attributable to meeting the 2006 targets since without these no capacity would exist, but that the variable costs of treatment above the levels required to meet 2006 targets are attributable to higher rates of treatment (despite the fact that commercially the plants would be operated at these higher rates). We examine the sensitivity of costs to this intermediate position.

The basic steps in preparing the estimates of costs (or cost savings) comprise:

- Step 1: Identify unit costs for different treatment options from the national cases and the technical review – these are essentially the average costs of each option and are usually expressed in terms of euros per tonne treated. This includes the costs of landfill which is avoided (ie a cost saving) as recycling/recovery targets increase.
- Step 2: Construct cost scenarios to capture the uncertainty and range in unit costs – three cost scenarios (high, medium, low) have been constructed to indicate the maximum range after combining both costs and avoided costs.
- Step 3: Examine the costs per treatment option per ELV when meeting 2006 recycling / recovery targets, taking into account the estimated weight and composition of ELVs in 2006, for each of the three cost scenarios compared to a standard ‘market plus depollution’ baseline.
- Step 4: Examine the costs per treatment option per ELV when meeting higher targets, taking into account the estimated weight and composition of ELVs in 2015, for each of the three cost scenarios compared to a standard ‘market plus depollution’ baseline.
- Step 5: Compare the costs per treatment option used to secure higher targets with the costs to meet 2006 targets to establish the additional cost per ELV, for each of the three cost scenarios.
- Step 6: Assess the effects on costs of excluding fixed costs, and examining other changes in cost assumptions (reflecting possible changes in the period to 2015).

6.2 Unit Costs of ELV Treatment Options

The review of technical options and the related costs, summarised above, and described in more detail in Annexes 2 and 3, has enabled the identification of the types and scale of cost for treating materials from ELVs in line with the targets set out in the Directive. These are summarised in Table 6.1.

Table 6.1: Unit Costs Used in Cost Assessment (constant 2005 prices)

Treatment Option	2006		2015	
	euro / tonne	euro/kg	euro / tonne	euro/kg
Landfill[1]:			+10%	
Low	35	0.04	39	0.04
Medium	65	0.07	72	0.07
High	115	0.12	127	0.13
Reuse / Shredding (income)[2]			0%	
Low	-60	-0.06	-60	-0.06
Medium	-80	-0.08	-80	-0.08
High	-100	-0.10	-100	-0.10
Dismantling and recycling (plastics, glass, textiles, small quantities only)[2]:			0%	
Low	200	0.20	200	0.20
Medium	300	0.30	300	0.30
High	1000	1.00	1000	1.00
Mechanical treatment of ASR[3]			-10%	
Low	20	0.02	18	0.02
Medium	75	0.08	68	0.07
High	100	0.10	90	0.09
Thermal treatment of ASR/Incineration with energy recovery[4]:			-10%	
Low	75	0.08	68	0.07
Medium	120	0.12	108	0.11
High	200	0.20	180	0.18

Note: All costs are gate prices. They therefore take account of net costs of processing, recycling and disposal of residues. Transport costs are excluded and assumed to be similar for different disposal routes.

1 Based on review of EU landfill costs in Annex 2

2 Based on country case studies, Annex 4

3 Based on gate price for mechanical separation based PSTs, Table 5.7 and Annex 3

4 Based on gate price for thermal treatment based PSTs, Table 5.7 and Annex 3

The unit costs for 2015, based on those identified for treatment options in 2005 have been adjusted to reflect real changes in costs that might be expected to occur over the next 10 years. In particular the costs reflect an assumption that landfill costs will increase in real terms by 10% over this period. This might be considered conservative in the light of the costs of compliance with the Landfill Directive and new landfill taxes and charges that might be introduced or increased over this period. The costs also reflect an assumption that the costs of new treatment methods as currently estimated will fall by 10% in real terms over this period. Again, in the light of past experience of reductions in the costs of new environmental technologies, as noted above, this might be considered conservative. All other costs and revenues are assumed to remain constant between 2006 and 2015.

It is also worth noting that the range in unit costs for the ASR – PST treatment options is indicative of the effects of scale economies on treatment plant. Thus the high end costs relate to smaller plants with fewer economies of scale. For those processes where scale economies can be calculated, the savings (per euro/tonne) for each additional 10,000 tonne of capacity range from 1 euro/tonne to 6.5 euro/tonne (Table 6.2).

Table 6.2: Indicative Effects of Scale Economies on Unit Costs of PST

Process	Unit Cost (Euro/tonne) Range (approx 100,000 to 200,000 tonne)	Indicative Cost Saving (Euro / tonne) per additional 10,000 tonne of capacity
Citron (Thermal Treatment)	100 – 200	6.5
Sicon (Mechanical Separation)	20 – 50	2.8
Twin-Rec (Thermal Treatment)	100 – 200	0.9
Reshment (Thermal Treatment)	75 – 140	4.0

6.2.1 Unit Income

The changes in composition and weight that are already taking place as reflected in the current EU vehicle fleet mean that there will be changes in the volumes of materials reused and shredded, as well as changes in the volume of material requiring treatment because of the diversion from landfill. Estimates of the possible range of income per tonne are drawn from the national case studies.

6.2.2 Preparing Cost Scenarios

Given the range in unit costs, and in particular the importance of the avoided landfill costs in calculating costs, we have estimated a range of costs presented as low, medium and high cost scenarios. To reflect the full range, low cost scenario uses the high landfill costs since these provide the greatest saving. Conversely the high cost scenario uses the low landfill costs since this provides the lowest savings. The medium cost scenario uses medium costs including medium landfill costs.

6.3 Indicative Costs of the Higher Targets

As noted at the beginning of this Section, the approach has been to define a baseline to represent current (largely market driven) practice and then to examine technical options capable of achieving the 2006 targets and then higher levels and their costs.

6.3.1 *Meeting the 2006 Targets*

The ELV Directive requires MS to achieve a reuse and recycling rate of at least 80% (by weight) and a reuse, recycling and recovery rate of at least 85% by 2006. These higher targets can either be met by increasing rates of dismantling, or by treatment of shredder residues, or by a combination of the two. The Dutch example, which is based on dismantling, will not necessarily be followed by other MS.

To assist in understanding the volume of treatment (kg) required by higher targets we have summarised these, and the associated diversion from landfill taking into account the change in vehicle weight:

	Base	2006	2015	Base - 2006	Base - 2015	2006-2015
Share of ELV Treated	80%	85%	95%			
ELV Weight (kg)	951	964	1025	13	74	61
Treated (RRR) (kg)	761	819	974	59	213	154
Landfilled (kg)	190	145	51	-45	-139	-93

Baseline

All Member States will be required to meet minimum environmental standards (depollution and treatment of tyres), which will increase rates of reuse, recycling and recovery to an estimated 80-81%. Table 5.4 above, which describes ELV treatment under "market-based plus depollution" practices therefore effectively provides a generalised baseline against which achievement of targets under other scenarios can be compared. All of the estimates of costs in Tables 6.3 to 6.9 below are based on this baseline. This avoids the need to use different baselines reflecting differences in current practice between Member States.

Dismantling Route

This route involves a similar approach to that employed by the Netherlands, including dismantling and recycling of larger and more easily removed items of glass, plastics and textiles. Because of the limited quantity of materials that can be cost effectively removed in this way, there is also likely to be some incineration and energy recovery from ASR, as has occurred in the Netherlands.

Table 6.3 summarises the additional recycling and recovery per ELV in moving from the baseline to the target. This involves additional reuse/recycling of 47kg per ELV and recovery of 10kg per ELV. The costs of dismantling and energy recovery are considerably higher than those for landfill and raise the net cost of treating an ELV by between 5 euro/ELV in the low cost scenario and 48 euro/ELV in the high cost scenario, with a medium cost of 13 euro/ELV.

Table 6.3: Cost of Meeting 2006 Target through Dismantling Route

Treatment		High cost scenario		Medium cost scenario		Low cost scenario	
	Kg/elv	Euro/kg	euro/elv	euro/kg	euro/elv	euro/kg	euro/elv
Change in:							
Dismantling and recycling	47.4	1.000	47.42	0.300	14.23	0.200	9.48
Thermal treatment of ASR	9.8	0.200	1.95	0.120	1.17	0.075	0.73
Avoided Disposal to Landfill	-44.2	0.035	-1.55	0.065	-2.87	0.115	-5.08

Additional Cost cf Baseline			47.83		12.53		5.13

Note that the avoided disposal to landfill is less than the level of treatment to reach the target by the amount ELVs increase in weight between the baseline and when 2006 targets are met – assumed to be 13 kg

Mechanical Treatment Route

Based on the unit costs, mechanical treatment of ASR compares more favourably with landfill in cost terms than dismantling. Table 6.4 presents estimates of the costs of meeting the 2006 targets by this means. Because treatment leaves an inorganic residue that needs to be landfilled or incinerated, accounting for approximately 25% of the ASR input, the quantity treated is one third greater than the rate of recycling achieved.

Table 6.4: Cost of Meeting 2006 Target through Mechanical Treatment Route

Treatment		High cost scenario		Medium cost scenario		Low cost scenario	
	Kg/elv	Euro/kg	euro/elv	euro/kg	euro/elv	euro/kg	Euro/elv
Change in:							
ASR mechanically treated	76.0	0.100	7.60	0.075	5.70	0.020	1.52
Material recycled from ASR treatment	*57.0*						
Landfilled residue from ASR treatment	*19.0*						
Avoided Disposal of ASR to Landfill	-44.1	0.035	-1.54	0.065	-2.87	0.115	-5.07
Additional Cost cf Baseline			6.05		2.83		-3.55

Note that the avoided disposal to landfill is less than the level of treatment to reach the target by the amount ELVs increase in weight between the baseline and when 2006 targets are met – assumed to be 13 kg

Table 6.4 suggests that meeting the 2006 targets will have a net cost in MS with high and medium cost scenarios of 6 euro/ELV and 3 euro/ELVs, respectively. There will be a net saving where the ASR treatment cost is less than the cost of landfill, which happens in MS with high costs of landfill. In this case there are net savings of 4 euro per ELV.

Thermal Treatment Route

Thermal treatment of ASR has higher costs than mechanical treatment techniques. Achieving the 85% target through thermal treatment of ASR has a net cost of between 1 and 14 euro per ELV (Table 6.5). Note that these techniques are not exclusively concerned with energy recovery (although that is the principal income stream) and enable increased rates of material recycling.

Table 6.5: Cost of Meeting 2006 Target through Thermal Treatment Route

Treatment		High cost scenario		Medium cost scenario		Low cost scenario	
	Kg/elv	euro/kg	euro/elv	euro/kg	euro/elv	euro/kg	euro/elv
Change in:							
ASR thermally treated	76.3	0.200	15.25	0.120	9.15	0.075	5.72
Energy recovered	26.2						

from ASR treatment							
Material recovered from AST treatment	31.0						
Landfilled residue from ASR treatment	19.1						
Avoided Disposal of ASR to Landfill	-44.2	0.035	-1.55	0.065	-2.87	0.115	-5.08
Additional Cost cf Baseline			13.70		6.28		0.64

Note that the avoided disposal to landfill is less than the level of treatment to reach the target by the amount ELVs increase in weight between the baseline and when 2006 targets are met – assumed to be 13 kg

6.3.2 Achieving Higher Rates of Recycling and Recovery

Increased rates of dismantling are unlikely to represent an option for meeting for achieving higher rates given the projected increases in related costs. Even marginal changes in the quantities of ELVs treated in this way are more expensive than methods focusing on ASR, while significant changes would be prohibitively costly. These costs have been recognised in the Netherlands as effectively limiting any further increases in dismantling. Germany has also recognised the costs of dismantling, specifically in relation to the removal of glass, and have granted an exemption to Annex 1 (4) requiring the removal of glass.

The analysis has therefore focused on the further use of PSTs to treat ASR and to achieve higher rates. The following estimates represent the costs relative to a current “market plus depollution” baseline, as used in the assessment of the costs of 2006. The resulting estimate of cost for achieving higher targets has to be compared with the previous estimates for 2006 to assess the additional cost of achieving higher rates than those set by the 2006 targets.

Mechanical Treatment Route

To illustrate the effect of higher rates on costs, Table 6.6 presents estimates of the cost of meeting the 2015 target through increased mechanical treatment. As in the case of meeting 2006 targets there are net cost savings to waste producers from using this technology where landfill costs exceed the cost of treatment. This is the case under the low and medium cost scenarios with cost savings of 1 to 21 euro/ELV. The net costs of treating an ELV under the high cost scenario is 10 euro.

Using this method of comparison, the additional costs of the higher rates of recyling / recovery compared to those required to meet the 2006 targets are between -17 and 4 euro per ELV using mechanical separation.

Table 6.6: Cost of Meeting the 2015 Target through Mechanical Treatment

Treatment		High cost scenario		Medium cost scenario		Low cost scenario	
	kg/elv	Euro/kg	euro/elv	euro/kg	euro/elv	euro/kg	euro/elv
Change in:							
Reuse	56	-0.060	-3.35	-0.080	-4.46	-0.100	-5.58
ASR mechanical treated	217	0.090	19.50	0.068	14.63	0.018	3.90
Material recycled from ASR treatment	*165*						

Landfilled residue from ASR treatment	*51*						
Energy recovery avoided	-10	0.068	-0.64	0.108	-1.03	0.180	-1.71
Avoided Disposal of ASR to Landfill	-138	0.039	-5.30	0.072	-9.85	0.127	-17.43
Additional Cost cf Baseline			10.21		-0.71		-20.81
Comparison with 2006 target			4.16		-3.54		-17.26

Note that the avoided disposal to landfill is less than the level of treatment to reach the target by the amount ELVs increase in weight between the baseline and 2015 – assumed to be 74 kg

6.3.2.1 Thermal Treatment Route

To illustrate the effect of using thermal treatment PST to achieve higher rates on costs Table 6.7 provides estimates of the cost of meeting the 2015 targets using thermal treatment and an element of mechanical recycling to ensure compliance with the higher recycling target. The net cost is between -6 euro per ELV in the low cost scenario situation and 28 euro in the high cost scenario.

Compared to the estimated costs of meeting the 2006 targets through the thermal treatment route, the estimated cost is between -6 and 15 euro per ELV.

Table 6.7: Cost of Meeting 2015 Target through Thermal Treatment

Treatment		High cost scenario		Medium cost scenario		Low cost scenario	
	kg/elv	Euro/kg	euro/elv	Euro/kg	euro/elv	euro/kg	euro/elv
Change in:							
Reuse	56	-0.060	-3.35	-0.080	-4.46	-0.100	-5.58
ASR thermally treated	165	0.180	29.76	0.108	17.86	0.0675	11.16
Energy recovered from ASR treatment	31						
Material recovered from ASR treatment	93						
ASR mechanically treated	43	0.090	3.84	0.068	2.88	0.018	0.77
Material recovered from ASR treatment	*32*						
Avoided disposal of ASR to landfill	*52*						
Landfilled residue from ASR treatment	-138	0.039	-5.30	0.072	-9.85	0.127	-17.43
Additional Cost cf Baseline			28.29		10.89		-5.50
Comparison with 2006 target			14.59		4.61		-6.14

Note that the avoided disposal to landfill is less than the level of treatment to reach the target by the amount ELVs increase in weight between the baseline and 2015 – assumed to be 74 kg

In summary the illustrations indicate the additional costs or cost savings per ELV of implementing a higher rate of recycling and recovery compared to the rates to meet the 2006 targets and are summarised in Table 6.8.

Table 6.8: Estimated Additional Cost of Achieving Higher Targets in 2015 (euro per ELV)

Options	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
Mechanical Separation	4.2	-3.5	-17.3
Mechanical & Thermal	14.6	4.6	-6.1

The additional cost can also be expressed on a euro per kg basis to assist in quantifying the effects of achieving different target rates. The effects of changes in targets can be approximated by estimating the treatment cost per kg (for given compositions of material subject to treatment) in 2015.

Because the basis of the cost estimates have been the same unit costs in both 2006 and 2015 (leaving aside the assumed changes in PST and landfill costs and changes in ELVs), then the marginal costs are unchanged between 2006 and 2005. These marginal costs per kg of landfill avoided are summarised in Table 6.9a.

Table 6.9a: Marginal Costs of Landfill Avoided (euro / kg) – Holding volumes / weights / cost changes constant

Options	Diversion (kg)	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
		Mechanical Separation		
Base – 2006	53	0.1	0.0	-0.1
Base – 2015	173	0.1	0.0	-0.1
2006 – 2015	119	0.1	0.0	-0.1
		Thermal Treatment		
Base – 2006	53	0.2	0.1	0.0
Base – 2015	173	0.2	0.1	0.0
2006 – 2015	119	0.2	0.1	0.0

Note: Vehicle weight 1,000 kg. Landfill volumes: Baseline: 209 kg; 2006: 166 kg; 2015: 50kg

The same calculation can be provided allowing for the changes in vehicle weight, composition and the assumed increase in real landfill costs and decrease in PST costs. These changes have a minor effect on the marginal costs per kg of landfill avoided (Table 6.9b).

Table 6.9b: Marginal Costs of Landfill Avoided (euro / kg) – adjusting for vehicle weight, composition and assumed changes in unit costs

Options	Diversion (kg)	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
		Mechanical Separation		
Base – 2006	44	0.1	0.1	-0.1
Base – 2015	138	0.1	0.0	-0.2

2006 – 2015	94	0.0	0.0	-0.2
		Thermal Treatment		
Base – 2006	44	0.3	0.1	0.0
Base – 2015	138	0.2	0.1	0.0
2006 – 2015	94	0.2	0.0	-0.1

Note: Vehicle weight: Baseline: 951 kg. 2006: 964kg; 2015: 1025kg

Landfill volumes: Baseline: 180 kg; 2006: 145 kg; 2015: 51kg

6.3.3 The Effects of Economies of Scale

The analysis of PSTs in the technical review identified the important effect of scale economies on the final gate fee of PST. The high cost scenario in part reflects costs arising from smaller plants and hence without the benefit of the scale economies. As a rule of thumb, based on the available information, plants in excess of 200 ktonnes of capacity would deliver unit costs closer to those used in the medium cost scenario. Of course larger plants may have greater difficult in sourcing a sufficient volume of treatable material, and in securing the necessary planning and environmental permits. Costs may be higher in small countries where the demand may only support smaller plants.

Note however that plants are built to take non-ASR material as well as ASR. The plants examined in the technical review typically operate an approximate mix of half ASR and half non-ASR; 200 ktonne plants will require approximately 100 ktonnes of ASR.

6.4 Meeting Higher Targets with Technology Installed to Meet 2006 Targets

The basic approach to the calculation of costs assumed that the treatment of ELVs would need to change and require more resources, to meet higher targets beyond those set for 2006. However, in most MS the achievement of the 2006 target of an overall rate of reuse, recycling and recovery of 85% has yet to be reached. The high costs of dismantling and the emergence of PSTs means that it is likely MS will seek to use PSTs to meet *both* the 2006 *and* higher 2015 targets.

If this is the case then, since the investment and operational costs will have been incurred to meet the 2006 target and, since plants are capable of achieving the 2015 targets, there are no additional investment costs of meeting higher targets. The PSTs are characterised by high fixed costs, which makes it commercially sensible to operate the plants at full capacity, achieving the maximum levels of treatment. As a result the plant operating in the counterfactual situation (ie without higher targets) is the same as the plant operating with higher targets. Since there is no change to the nature of the plant required by the higher targets, there is no additional cost attributable to higher targets. The costs of the PSTs are set against the achievement of the 2006 targets.

The cost analysis is based on the average (unit) costs of PSTs operating at full capacity, and hence includes both fixed and variable costs and has attributed capacity and associated costs of treatment to rates beyond 2006 target levels.

If Member States do use PSTs to meet 2006 targets, the correct approach to estimating costs of higher rates beyond 2006 target levels is to look at only variable costs. This approach seeks to recognise that there are costs of higher levels of treatment, which could be attributable to higher targets, but that since investment was made to reach the 2006 targets the fixed costs are not attributable.

If MS do use PSTs to meet 2006 targets the installed capacity will not need to change in order to achieve higher rates of recycling, plants will be operated at full capacity delivering the higher rates. The issue is one of attribution of cost rather than the need for new capacity to achieve higher rates.

Table 6.10 summarises the ratio of fixed and variable costs for the four technologies for which detailed cost data is available, for plant with an approximate throughput of 200 ktonnes. Smaller plants will have a higher share of fixed costs. Net variable costs vary between the PSTs but range from 22% to 53% of total costs per tonne, with three of the four technologies ranging from 42% to 53%.

Table 6.10: Summary of Available Fixed and Variable Costs for PST Plant (c200kt)

Cost Parameter (euro per tonne)	Sicon	Citron	Twin Rec	Reshment
Fixed Cost	11	71	44	37
Gross Variable Cost (GVC)	27	97	59	48
Sales	19	77	10	20
Net Variable Cost (GVC less sales)	8	20	49	28
Total Cost	19	91	93	65
Net Variable Cost as % of Total Cost	42%	22%	53%	43%

Source: Review of Technologies, Knibb Gormezano and Partners for the ACEA, "Recycling Infrastructure & Post Shredder Technologies"

Assuming an average of 40% then the unit costs would be 40% lower than those applied in the analysis presented above. Under these conditions the marginal cost per kg of landfill avoided between 2006 and 2015 levels falls to zero even in the high cost scenario (Table 6.11).

Table 6.11: Marginal Costs of Landfill Avoided (euro / kg) – Based on Variable Costs

Options	Diversion (kg)	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
		Mechanical Separation		
Base – 2006	44	0.0	0.0	-0.1
Base – 2015	138	0.0	-0.1	-0.2

2006 – 2015	94	0.0	-0.1	-0.2
		Thermal Treatment		
Base – 2006	44	0.1	0.0	-0.1
Base – 2015	138	0.1	0.0	-0.1
2006 – 2015	94	0.0	0.0	-0.1

Note: Vehicle weight: Baseline: 951 kg. 2006: 964kg; 2015: 1025kg

Landfill volumes: Baseline: 180 kg; 2006: 145 kg; 2015: 51kg

In terms of the additional cost per ELV the effect is to reduce costs to 4 euro per ELV in the high cost scenario with thermal treatment, and cost savings in all other scenarios. Cost savings, in the low cost scenario with mechanical separation are 18 euro per ELV.

6.5 Sensitivity Analysis

The estimates of costs are based on a range of available unit cost data combined with data and scenarios on future ELV weight and materials and their treatment. The different cost scenarios and examination of different treatment scenarios and treatment options is already designed to take full account of the uncertainties implicit in an exercise designed to assess costs occurring in ten years time.

The analysis has not however introduced any distinction between different MS and in particular the differences in costs between new and old MS. The new MS are less likely to face the low cost scenario because of the limited savings that would accrue from the diversion from landfill because landfill costs are relatively low. However, under the provisions required by the environmental acquis NMS will be required to replace non-compliant low cost landfill capacity, leading to real increases in landfill costs. We have conservatively assumed that landfill costs rise slowly over the next decade in real terms, by only 10%. If landfill costs were to increase by say 25%, the costs of higher targets would decrease, by between 1 and 2 euro per ELV.

We have also examined the effects of a more rapid reduction in the gate fees for PST from current estimates. We have conservatively assumed that these costs would decrease by 10% in real terms over the decade. If costs were to decrease by say 50% over the period (a reduction not inconsistent with the experience of other new environmental technologies), the costs would decrease by up to 15 euro per ELV for the high cost scenario using thermal treatment and by 8 euro for mechanical separation. In this case even with just the modest rise of 10% in landfill costs all options provide cost savings.

New MS are also likely, because of the lower volumes of material arising, to require smaller PST plants, and hence to avoid the benefits achievable from scale economies. We have already examined this effect. The economies of scale (of say a 200k tonne plant compared to a 100 ktonne plant) provides a cost saving of between 10% and 100% depending on the technology with the likely impact towards the higher end of this range..

6.6 Issues of Feasibility

The estimation of costs of achieving the target rates provides the basis of a number of conclusions relating to the implied technical response to the targets. In drawing these conclusions it is important to stress that they stem in large part from the available information on the capacity and cost of post-shredder technologies. The estimates have been based on the best current information on PSTs.

6.6.1 Dismantling as the Main Treatment Option

The analysis of the costs of dismantling indicates that this represents by far the most costly approach. The sharp increase in effort required to secure the level of dismantling required beyond the removal of parts for commercial sale and of parts that can be easily removed (such as bumpers) means that alternative treatment options are more likely to offer a cost competitive response. Even in the low cost scenario dismantling is five times the cost of thermal treatment, the next most expensive option to secure the 2006 target of 85%.

This cost has been recognised with some MS (eg NL) investing in alternative PST options and other MS (Germany) removing the need for certain items to be dismantled (eg glass).

6.6.2 The Effectiveness of PSTs in Achieving Higher Targets

If dismantling is used only as a limited option, then the issue is whether any other option is capable of achieving higher rates of recycling / recovery up to say 95% when combined with the market and depollution baseline. The review of PST indicates that based on available information the technologies are capable, when combined with the current 'market and depollution' levels of achieving these higher rates. Perhaps surprisingly, information on three of the four mechanical separation technologies suggests that they are able to achieve a 95% rate through recycling, with no reliance on recovery.

6.6.3 The Effectiveness of Achieving a Higher Recycling Rate

The feasibility of achieving higher rates of recycling up to 95% is suggested by seven of the eight PSTs. The thermal treatment processes, although concerned to extract the calorific value of the waste stream would seek to recycle the residual metal fractions remaining after the shredder process, and depending on the process, the glass fraction.

Since the PSTs appear capable of ensuring treatment conforming with the proposed 2015 targets, the question is whether there are strong environmental arguments for recycling compared to recovery which would justify the identification of a preferred route. The relative environmental benefits of different treatment options are examined in the next section.

Excluding Sicon, the costs of the two processes are not dissimilar. However if the Sicon process can be demonstrated to operate as indicated at the costs suggested then this may limit interest in other PST. The planned investment by ARN of the NL to test the Sicon process in an industrial scale plant will be important in establishing the feasibility of the process in relation to the two target rates (Box 6.1). However, the delay before the technology is proven may allow market advances in other PST.

Box 6.1: Plan for Sicon Separation Plant in Netherlands

In collaboration with Volkswagen AG and the German engineering firm SICON GmbH, ARN is preparing to open a plant with a capacity of 100 ktonnes a year in 2007. Volkswagen and Sicon are providing the technology; the investment will come from ARN and its partners in the waste industry.

The main products recovered from the shredder waste are plastics, fibres and mineral fractions and the last scraps of metal. Selling the ferrous and non-ferrous metals and the pure plastic fraction does not represent a problem. Pilot projects are currently underway with a number of customers to investigate whether the mineral, fibre and mixed plastic fractions are suitable for further recycling.

ARN has asked the Ministry of Spatial Planning, Housing and the Environment (VROM) to review the current policy on landfilling since it is based on the absence of suitable technology for the recycling of shredder waste. Once the separation plant has been built the technology will exist.

The Ministry has meanwhile stated its intention to increase the environmental tax payable on shredder waste from 1 January 2008. A landfill ban will only enter into force on 1 January 2009 to give the market for the remaining shredder waste the time to create sufficient recycling capacity.

The ministry has also said that the Management of End-of-Life Vehicles Decree will be amended to bring forward the date by which the 95% target has to be met. This amendment is conditional on the new separation plant actually being built.

Source: Auto Recycling Nederland (ARN), Press Release, Amsterdam, 26 October 2005, and contact with Janet Kes (ARN) 08 May 2006

6.6.4 *The Constraints on the Development of the PST*

The potential use of PST to secure the proposed targets requires some consideration of the possible constraints, other than the technological development itself. There are a number of factors to consider:

- Importance of market certainty – EU approval of an overall 95% recycling / recovery rate would underpin the required investment and provide the necessary confidence in the demand for the technology.
- Environmental permitting – major plants are likely to be subject to IPPC regulations (especially where thermal treatment processes are used). This will impose permitting costs. In addition plants will require planning approvals as industrial plant. However, from a compliance monitoring perspective PSTs provide scope to increase the efficiency of the regulation of the sector compared with the present more fragmented system.
- Competition with landfill – until the targets become legally enforceable landfill operators will be able to compete for the wastes. This will be a particular problem in low landfill cost countries. The NL proposal to require disposal other than at a landfill once alternative capacity comes on stream could ensure both earlier investment and add additional security to investors. In addition existing policies to promote full cost recovery (including account of the environmental

externalities of landfill sites such as emissions, noise, odour and contamination) would also support investment.

- Access to investment funds – the investment required is significant. In the technologies reviewed the investment cost for a 200kt plant ranges from approximately 11 million euro (Sicon) to 90 million euro (Citron). However, the major shredder companies are already significant businesses, and would, depending on contractual agreements with producers, be supported by the new take back and treatment provisions which are currently being negotiated with the vehicle producers. The high initial investment cost is therefore unlikely to constrain development.
- Lead times – long lead times would increase investment risks. It is clear from the technical review that further technological development is required to prove the PSTs at industrial scale. In the opinion of the Stakeholder Group report, however, lead-times could be as little as five years. Clearly, the earlier the confirmation of the required targets the sooner investment risks can be reduced. To the extent that PST investment is made as a means of meeting the 2006 targets (or simply as a commercial venture) there is likely to be less delay. To achieve radical cost reductions through technical change and the introduction new technologies will require longer lead times.

6.6.5 Additional Benefits from the PST

The investment in PST as a solution to the treatment of ASR and the achievement of the 95% target would also potentially have other technical benefits. The first relates to the significant monitoring and measurement problem raised by having to ensure compliance with target rates of recycling and recovery, based on measurements down to tens of kilograms of materials. Fragmentary and dispersed treatment options would exacerbate the problem. However, PSTs would provide an easier solution.

A second benefit is that there may be economies of scale for the development of technologies and facilities for the treatment of waste electric or electronicequipment required (WEEE) under the WEEE Directive. PSTs provide capacity to comply with the WEEE as well as the ELV Directive. Technology which reduces costs for ELV treatment will also reduce costs for WEEE treatment.

7 ECONOMIC IMPACT

7.1 ELV and Material Arisings

This section provides an estimate of the number of ELVs requiring treatment in the EU and projected to require treatment in 2015.

Annex 2 presents data on the stock of vehicles, and the reported number of vehicle de-registrations and ELVs treated in the EU. There is also trade between MS and third countries that has to be taken into account in both second hand vehicles and body shells.

Based on the available data, and estimates where there are gaps in the data, our best estimate of the number of ELVs requiring treatment in the EU25 each year is 10.5 million, (Table 7.1).

Table 7.1: Estimated Number of ELVs Requiring Treatment in EU25 (2004)

	Vehicles (000)
Deregistered in EU15	11,296
Deregistered in new MS	1,342
Deregistered in EU25 (1)	*12,638*
Of which:	
Exported to another MS (2)	2,068
Exported outside EU (3)	995
ELVs requiring treatment in EU:	
Minimum *estimate = (1) – (2) – (3)*	9,575
Maximum *estimate = (1) – (3)*	11,643
Midpoint	**10,609**

The lower estimate of 9.6 million vehicles requiring treatment in the EU is based on an assumption that vehicles that are exported are deregistered twice, once in the country exporting them and once in the country in which they end their life. Thus intra EU trade is deducted from deregistrations to estimate the number of ELVs requiring treatment. In practice many vehicles are likely to be traded and treated as ELVs without being re-registered. This presumably explains why, for example, the number of vehicles treated in Spain exceeds the number of deregistrations.

The upper estimate of 11.6 million ELVs is based on the assumption that each vehicle is only deregistered once, assuming it is not re-registered in the importing Member State. The reality might be expected to lie somewhere between these two positions.

This estimate can be compared with the stock of of vehicles in the EU. The deregistration statistics suggest that only 5.3% of ELVs are deregistered each year, reflecting continuing growth in the vehicle stock and hence a relatively low average vehicle age. If the number of vehicles in the EU were to stabilise, an average vehicle life of 12.5 years would suggest that 8% of vehicles would reach the end of their life each year.

The number of ELV deregistrations is increasing as the stock of vehicles grows. The rate of growth of deregistrations is clearly a function of the age of the vehicle stock as well as the number of vehicles in use. In some Member States, this growth is particularly rapid. For example, Poland expects the number of annual deregistrations to increase from 250,000 between 1997 and 2000 to 540,000 in 2006; 700,000 in 2010; 800,000 in 2012 and 950,000 in 2014 (figures from Polish National Waste Management Plan).

Based on an average annual growth rate in the EU stock of vehicles of 2.4% and an assumed average life of vehicles of 12 years, the stock of ELVs requiring treatment in 2006 and in 2015 is 11.1 million and 13.8 million respectively.

7.2 Gross Costs of the Directive

The gross costs of the increase in targets in 2015 is calculated by multiplying the additional cost per ELV, of meeting the overall target of 95% recycling or recovery with the number of ELVs requiring treatment. Note that the additional costs as calculated in the previous section take into account both the projected increase in average vehicle weight and the change in vehicle composition.

The estimated cost is indicated in Table 7.2. This shows a range, depending on the choice of technical option and the projected level of landfill costs from a net cost saving of 240 million euro to a cost of 200 million euro.

Table 7.2: Estimated Annual Gross Cost of Higher Targets in 2015 (million euro)

Options	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
Mechanical Separation	57.4	-48.9	-238.2
Mechanical & Thermal	201.4	63.6	-84.7

Assessing the costs of higher targets when based on the variable costs of PST, (attributing the fixed costs to meeting 2006 targets), indicates (Table 7.3) that there are only gross costs in the high cost scenario with thermal treatment. In all other cost scenarios the financial benefits of avoided landfill costs is greater than the treatment costs.

Table 7.3: Estimated Annual Gross Cost of Higher Targets in 2015 (million euro)

Options	High Cost Scenario	Medium Cost Scenario	Low Cost Scenario
Mechanical Separation	-35.9	-114.3	-243.7
Mechanical & Thermal	49.4	-32.3	-136.1

7.3 Incidence of Costs

The ELV Directive seeks to encourage MS to promote producer responsibility. To this end the Directive requires that there be no charge payable by the final vehicle owner when the vehicle is deregistered, and that any costs associated with the subsequent treatment of the ELV are absorbed by the vehicle producers after taking into account the value of the ELV. This in turn suggests that producers (or government on their behalf) would recover any additional costs through a premium on the sale of new vehicles, since all producers selling into the EU market would face similar charges.

The additional costs of the higher targets are estimated to be between -17 euro (ie a saving on current treatment costs) and 15 euro. As a share of the price of a new vehicle the additional costs are very small (perhaps 0.2% in the case of the highest costs on a small vehicle). As a share of the life cycle fuel cost of operating a vehicle, the additional costs are also small, approximately 0.3% in the case of the highest treatment costs[56]. These costs will either, depending on MS decisions on the implementation of producer responsibility, be absorbed by the producer or shared with consumers.

It is also worth noting that the current levels of treatment are financed out of the positive resale value of parts and scrap metal. The combination of increased vehicle size and changes in material composition result in ELVs comprising an additional 11 kg of ferrous (steel) and 15 kg of non-ferrous (aluminium) material in 2015 compared to 2006. This material is worth some 30 euro per ELV at current scrap metal prices and is well in excess of the costs as estimated even under the high cost scenario. This suggests that any additional costs of higher targets will be borne by the last user receiving less for their ELV.

The study has also examined the possible transfer of costs to those Member States who import second hand vehicles from other MS, and in particular to the new east and central Europe MS. Analysis of the data on second hand car imports indicates that they represent about 1% of the vehicle stock in the EU25, but about 3% in the eight east and central Europe MS. There is therefore some evidence that treatment capacity will be disproportionately located in the new MS. To the extent that there are additional costs not covered by the value of an ELV, producer responsibility should mean the costs are borne by the purchaser with costs effectively transferred back to the MS where the car was purchased. This assumes that the national authorities are able to enforce producer responsibility.

7.4 Effects on Operators

7.4.1 Vehicle Producers

The effects of implementing the higher targets on vehicle producers relate firstly to the additional costs incurred under the free take back provisions; and the requirements which MS may introduce to make producers responsible for the costs of treating ELVs received under the provisions over and above the residual value of ELVs. In the case where residual values after depollution costs are close to zero then all the additional costs will potentially (subject to MS regulations covering producer responsibility and the detail of contracts between the producers and treatment sector) borne by the vehicle producers.

The effects of the targets on vehicle design have been considered; with little evidence that provisions have influenced design, not least because from a life-cycle perspective of a vehicle's manufacture, use and disposal, the disposal phase represents a minor contribution to life cycle impacts requiring greater focus on other phases, especially the use phase. It has been suggested in personal communication that producers have sought in some components (for example in door panels) to retain the use of metals instead of other materials to improve the residual value of ELVs and hence to contribute to reducing their costs of the take back scheme.

Given the average life of vehicles, design changes not yet implemented will not effect the weight or composition of ELVs arising in 2015. However, it seems reasonable to consider that under the much closer collaborations between producers and the treatment sector

[56] Based on the lifetime fuel costs of a vehicle of 5,700 euro, based on 150,000km travelled over the life of a vehicle, average vehicle weight of 1,000kg average fuel consumption of .38 litres/100 kg * 100km - (p14 LIRECAR) and average EU25 unleaded petrol cost of 1.23 euro per litre

required by the take back provisions that future opportunities will be identified to minimise the costs of the treatment processes; and hence influence design over the coming years.

7.4.2 Vehicle Treatment Sector

ELVs may be received by a large number of organised collection points or small operators, including scrap yards, dismantling businesses, salvage operators and secondary metals businesses. In contrast, shredding plants are large, capital intensive operations and are relatively few in number. For example. in the UK, ELVs are collected by 2,500 small scrapyards, dismantlers and recycling businesses, while there are some 37 shredders, with around 70% of capacity controlled by two companies. The German network consists of approximately 15,000 reception/collection points (e.g. dealerships and garages, which then pass on ELVs to recycling businesses), 1,200 dismantling/recycling businesses and 41 shredder plants. Poland has an estimated 1,500 scrapyards, over half of which are still unauthorised in 2005, but only four shredding facilities.

Table 7.3 from the stakeholder report[57] provides estimates of the number of authorised treatment facilities and shredders across the EU. It is estimated that there are nearly 8,000 ATFs but only 232 shredders. ATFs are estimated to process an average of 4,300 ELVs each while the average throughput of a shredder is some 34,000 vehicles.

Table 7.3: Outline of the Treatment Sector

Member State	No. of ATFs	No of ATFs certified	*No. treated ELVs per ATF*	No. of shredders	*No. treated ELVs per shredder (000)*
Austria	200	200	620	6	21
Belgium	48	48	*1,917*	12	*8*
Cyprus	1	?	*?*	0	*?*
Czech Republic	80-100	?	*?*	3	*?*
Germany	1,178	1,178	*1,019*	41	*29*
Denmark	210	210	*381*	13	*6*
Spain	540	501	*1,852*	22	*45*
Estonia	70	?	*214*	1	*15*
Greece	4	?	*5,000*	4	*5*
France	1,000	420	*1,300*	42	*31*
Finland	60	30	*1,483*	2	*45*
Hungary	150	?	*?*	2	*?*
Italy	1,800	314	*508*	18	*51*
Ireland	35	35	*3,714*	2	*65*
Luxembourg	2	1	*4,500*	0	*n.a.*
Latvia	161	?	*311*	1	*50*
Lithuania	43	?	*465*	1	*20*
Malta		?	*?*	0	*?*
Netherlands	500	500	*544*	11	*25*
Portugal	8	1	*6,500*	2	*26*
Poland	670	?	*119*	4	*27*
Sweden	370	120	*641*	7	*34*
Slovenia	20	?	*?*	1	*?*
Slovakia	30	?	*?*	1	*?*
UK	732	732	*2,883*	37	*57*
EU	**7,922**	**4,290**	**1,788**	**232**	***34***

Source: Stakeholder Report, 2005

[57] Stakeholder Consultation on the Review of the 2015-Targets on Reuse, Recovery and Recycling of End of Life Vehicles, 2005

The effects of the higher targets on the treatment sector obviously depends on the treatment scenarios; and in particular increased use of dismantling processes or the use of PSTs. Given the much higher cost of dismantling and the growing investment in PST we have focused on the effects of the introduction of PST plants and the effects of the diversion of ASR from landfill.

Based on a target of 95%, an average ELV weight of 1025kg and a baseline level of reuse, recycling and recovery of 80%, PSTs will need to treat some 150kg of ASR per ELV which would otherwise be landfilled. Based on an estimated 14m ELVs requiring treatment in 2015, this approximates to some 2.1 million tonnes of ASR requiring treatment. Based on an average PST capacity of 200 ktonnes, and a 50% use of ASR per PST, this volume would require the addition of 21 PST plants across the EU.

In practice smaller plants will be required in some MS and in some locations, with the consequent need for a greater number of plants.

This additional treatment activity will need to be absorbed by the treatment sector, with considerable scope for vertical integration between shredding and post-shredding activity, which may force some rationalisation into fewer larger plants. The present numbers of shredding facilities might therefore be expected to decline in number but increase in average throughput of ELVs.

The gross effect of the additional treatment capacity on investment and employment can be calculated using the information obtained on the emerging PST. This data suggests that investment costs of PST range from 11 million euro (Sicon) to 90 million euro per 200 ktonnes plant. The total investment for the 21 plants therefore ranges from 230 million euro to 1.9 billion euro; with ASR attributed investment of 115m euro to 950m euro.

The employment effect can be calculated using the average labour costs per tonne identified in the technology review. This indicates a cost of approximately 7 euro per tonne treated per year and hence payment for labour of 14.7 million euro. Based on an average labour cost per worker of 34,400 euro per year, the volume of ASR treated will generate approximately 400 jobs. However, there will be displacement of employment from the landfill sector associated with the reduced volumes of ASR. Given the capital intensive nature of the PST process the displacement may be greater than 100%, i.e. there will be a net loss of employment; partly reflecting the increased efficiency of treatment, which in turn will provide incentives for future employment growth.

7.5 Effects on European Competitiveness

The effects of the higher targets on European competitiveness has been considered from the perspective of the effects on international markets and producers and on resource efficiency.

7.5.1 International Approaches to ELV Treatment

If producers pass on additional costs to EU consumers then little competitive impact can be envisaged. Since imported vehicles would be subject to the provisions of the Directive there is no advantage to producers operating outside the EU. Indeed, to the extent that non-EU producers would find it more difficult to put in place the necessary take back and treatment systems, the Directive may increase barriers to entry into the EU market.

The study has examined the extent to which there are provisions similar to the ELV Directive in other vehicle markets, which would effectively reduce any competitive disadvantage to producers operating in the EU market and facing additional costs, where costs are not passed on.

In summary, reviews of activity in the US, Japan and Australia (see national case studies) indicate that policies to improve the treatment of ELVs and to reduce volumes of material disposed as waste are progressing in all three countries.

7.5.1.1 Japan

Japan has the strongest set of policies in the three countries reviewed (driven by the scarcity of landfill capacity). These closely mirror the ELV directive, coming into force in the beginning of 2005. The law requires 95% rates of recycling of ASR by 2015. The provisions require consumers to pay a recycling charge, levied on new cars and on used cars at the point of inspection to finance the associated costs. Revenues are collected in a fund and distributed to manufactures / importers, when treatment has been certified, who then reimburse the treatment operators for costs.

The Automobile Shredder Residue Recycling Promotion Team (Art)

A new alliance for ASR recycling called the Automobile Shredder Residue Recycling Promotion Team (ART) has been formed in Japan between Suzuki, Nissan, Nissan Diesel, Fuji Heavy Industries, Mazda, Mitsubishi and Mitsubishi Fuso Truck & Bus. Nissan has been selected as team leader. The aim of ART is to take responsibility for activities ranging from ASR recovery to recycling and disposal. This will involve working with a number of commercial operators with recycling know-how to set ASR recovery criteria, assign recovery locations, and examine recycling methods. The ART expects to benefit from efficiencies by outsourcing the work involved in shipping management, recycler/final disposal handler management and payment for recycling and disposal fees to businesses and other entities that have the relevant know-how. The formation of the new alliance enables the carmakers to share their accumulated knowledge and experiences of recycling. It will also allow commercial entities to contribute their know-how and ensure transparency in all activities and aim to minimize costs through streamlining those activities.

The collection of ASR from all companies is also expected to generate economies of scale. There is scope for benefits exceeding those that would accrue if each vehicle manufacturer recovered and recycled ASR independently.

Current recycling methods for shredder residue usually involve high-temperature processing, which allows the recovery of energy and metals. The remaining residue is used for road surfacing and concrete reinforcement materials. Efficient high-temperature processing, which allows the quantity of shredder residue to be reduced, is an extremely complex technology. The object is improve technical designs for treating ASR.

Nissan were selected as team leaders because they have made significant progress in recycling ASR. From 1997, Nissan has worked on the recycling of ASR. Nissan rebuilt part of their waste incineration facilities at the Oppama plant, tested and achieved solutions to technical problems and started energy recovery of ASR towards the end of 2003. This was the first time any carmaker had used existing incineration facilities at its own plant to process ASR. The vapour generated by the process will be used for heating in the paint process and elsewhere along the production line, making the plant a leader in energy conservation. The technology and know-how adopted by Nissan could be applied at other waste incinerators, and the information is being shared as part of the ART.

Sources: www.theautochannel.com. Mazda, Social and Environmental Report, 2004. Nissan, Sustainability Report, 2005.

7.5.1.2 US

Reviews[58] suggest that ELV developments in Europe and Japan are having direct and indirect impacts on US manufacturers. US manufacturers are required to comply with regulations in their European/Japanese manufacturing and sales operations. There is some reported evidence that US manufacturers are using their experiences in Europe and Japan to increase domestic car recycling in the US, and are trying to limit the need for specific regulations in the US by demonstrating voluntary progress in ELV recycling. There is also likely to be competitive market pressures from European/Japanese manufacturers operating in the US, who will be stressing the importance of recycling in their own marketing.

US manufacturers are researching, and taking steps towards, increasing ELV recycling rates on a voluntary basis. General Motors, Ford and Chrysler have formed the US Council for Automotive Research (USCAR), a key objective of which is to promote and conduct research required for the technology to recover, reuse and dispose of materials from ELVs. All US car manufacturers have developed lists of restrictions on materials that are now used as specifications for their suppliers to restrict or exclude specific substances.

Ford Bumper Recycling

Ford has introduced worldwide recycling guidelines and is seeking to increase the use of recycled materials and the recyclability of materials used in the manufacturing process. Ford has a target to achieve a minimum 25% post-consumer recycled content of the plastic materials used in Ford cars. To assist Ford in achieving this target they have been running a bumper recycling programme in the US since 1993 (General Motors has also been running a similar programme), initially looking at recycling bumpers into housing for headlights, and using recyclates from bumpers in new bumpers.

The bumpers are collected through a network of dismantlers who are paid $4 per bumper. These are recycled, with more than 3,000 tonnes of plastic per year processed. The recycled material is used by Ford to manufacture new bumpers, and anything not used by Ford is sold to other manufacturers. The recycled material is sold at a 25-30% cost saving compared to virgin material. This programme is diverting almost 500,000 bumpers from landfill each year. Ford estimates that the bumper recycling programme will save about $1 million per annum, and improves Ford's image with marketing benefits.

7.5.1.3 Australia

The review[59] indicates that there is no equivalent policy to the ELV Directive. The current lack of any formal ELV deregistration requirements is considered by government to be contributing to the costs and inefficiencies currently experienced in collecting and treating ELVs, and there is growing support for a system whereby a requirement would be placed on the last owner of each vehicle to formally deregister ELVs, leading to appropriate de-pollution and parts recycling at accredited dismantlers, in line with the European legislation. However, the Federal Chamber of Automobile Industries and their members indicated opposition to any imposed reuse and recycling targets on Australian manufacturers. They claim that the cost impact on local manufacturers would be unwarranted at the present time given the different conditions in Australia compared to Europe and Japan and in particular the lower landfill charges in Australia.

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[58] Zoboli et al, Regulation and Innovation in the Area of End-of-Life Vehicles, March 2000; Bandivadekar et al, A Model for Material Flows and Economic Exchanges Within the UK Automotive Life Cycle Chain, 2004

[59] Sources: www.apraa.com, www.pacia.org.au, Department of the Environment and Heritage, *Environmental Impact of End-of-Life Vehicles: An Information Paper,* 2002

Some progress has been made towards encouraging ELV recycling through informal encouragement of recyclers and dismantlers. A joint project between the Environment and Heritage Department and APRAA has produced guide booklets on waste oil recycling, which were sent to recyclers and dismantlers throughout Australia during 2003. They are also encouraging recyclers to prepare their own environmental action plan, highlighting the efficiency benefits and improved reputation amongst councils, environmental protection agencies, customers and staff, of any operators taking actions to protect the environment. There is also a current project being undertaken by the Plastics and Chemicals Industries Association (PACIA) looking at the potential for recycling automotive plastics.

7.5.1.4 Conclusions

The EC in the European Competitiveness Report 2004, suggest that because the ELV Directive concerns all cars sold in Europe no cost disadvantage will arise for European manufacturers compared to non-European ones. However, the legislation may create barriers for any vehicle manufacturers for whom the costs associated with meeting the Directive outweigh the potential reward from doing so, for example manufacturers for whom Europe and/or Japan only account for a small proportion of their business but who nevertheless have to organise take back and treatment systems.

Moreover, the argument is advanced that the proximity of the European automotive industry with car-recycling firms constitutes a competitive advantage, considering that non-European manufacturers have not as yet established as dense a network of dealers for managing the ELV take back and treatment. It is evident from the national reviews above that greater co-operation between producers and the treatment industry is increasing, especially in Japan; with increasing capacity provided for innovation in vehicle design to reduce ELV treatment costs.

The multinational nature of the vehicle manufacturing industry would suggest that although the new legislation in Europe and Japan is likely to cause barriers to entry to these markets, in the short term, it is unlikely to have a significant impact on competitiveness amongst the major vehicle manufacturers.

7.5.2 Resource Efficiency

The response to higher targets using PSTs has the potential to improve European competitiveness through the greater resource efficiency implied by the cost savings in the low and medium cost scenarios. In gross terms these resource savings have been estimated to be up to 240 million euro per year. This response therefore makes useful contribution to the wider EU goals of greater resource efficiency.

Improvements in resource efficiency should have a beneficial effect on the EU trade balance. In addition the development of new technologies to achieve higher rates of recycling / recovery provides future opportunities for exports.

8 ENVIRONMENTAL IMPACT

In this chapter we analyse the environmental impacts and benefits of recycling and recovery options. It is composed of 3 sections:

- Section 8.1 focuses on plastic resins
- Section 8.2 deals with 2015 targets with the analysis of the scenarios described here above (see 5.6 Table 5.11)
- Section 8.3 is similar to section 8.2 except that it is for 2006 targets (as explained in §0) and analyses the scenarios described in section 5.6 Table 5.10

As plastics are the main material involved in the scenarios to reach higher recycling and recovery targets than those established for 2006, it was relevant to first focus on them.

8.1 Environmental impacts & benefits associated with different treatments of plastics

8.1.1 Objective of the analysis

Question analysed: **what are the environmental impacts and benefits of different alternative treatments for key plastic fractions?**

8.1.2 Scope of the analysis

Pieces / resins and treatment alternatives considered

Table 8.1: Pieces / resins analysed in this section

	kg/ELV	Mechanical recycling	Energy recovery in MSW incineration plant	Energy recovery in cement kiln	Recovery in blast furnace	Recovery during syngas production	Landfill
PUR (seat cushion)	2.4 (Fraunhofer) 1.20 (APME)	X (3 types)	X (2 types)	X (2 types)	X (2 types)	X (2 types)	X (2 types)
PP/EPDM (bumper)	4.93	X	X (3 types)	X (2 types)	X (2 types)	X (3 types)	X
PP (bumper and air duct)	0.95 in air duct and 3.14 in bumper	X	X	X	X	X	X
PA-6.6 GF (hubcap)	0.474	X	X	X	X	X (2 types)	X
PA (intake manifold)	0.72	X	X	X	X	X	X
PVC, ABS, PUR, PP-TV (dashboard)	5.054	X (2 types)	X	X	X	X	X
PE (wash fluid tank and lid)	0.43	X	X	X	X	X	X
ABS (mirror housing)	0.27	X	X	X	X	X	X
PC (headlamp lens)	0.30	X	X	X	X	X	X

Remark: as explained hereafter, the resins analysed in this report are those for which LCA data are available in the literature.

Remark: in the rest of the report, for readability purpose, results are presented per resin, without systematically mentioning the ELV piece concerned. However, **some of these results may be piece-dependent. Thus results have to be considered valid for the concerned ELV piece**. It may not be correct to extrapolate them to other resins.

Limits of the scope of this exercise

As illustrated in the table below, cars contain up to 140-150 kg of plastics distributed in various pieces. More than 15 different resins are involved and a plastic piece is rarely made out of one pure resin but out of a mix of resins.

Table 8.2: Plastics by type and application, per car for 2005

PART	MAIN PLASTICS TYPE	WEIGHT IN AVERAGE CAR (kg)
BUMPERS	PP	**10.4**
SEATS	PUR, PP, PA, PVC, ABS	**18.4**
COCKPIT	PP, SMA, ABS, PC, PVC, PUR	**21.3**
FUEL SYSTEMS	PE, POM, PA	**8.6**
BODY (including body panels)	PP, PPE, UP	**10.8**
UNDER THE BONNET COMPONENTS	PA, PP, PBT	**13.8**
INTERIOR TRIM	PP, ABS, POM, PVC, PUR	**31**
ELECTRICAL COMPONENTS	PP, PVC, PA, PBT, PE	**10.3**
EXTERIOR TRIM	ABS, PA, PP, PBT,ASA	**5.1**
LIGHTING	PP,PC, ABS, PMMA, UP	**5.6**
UPHOLSTERY	PUR, PP, PVC	**6.8**
OTHER RESERVOIRS	PP, PE, PA	**1.5**
TOTAL		**143.4**

Source: PlasticsEurope, 2005

Reliable LCA data in literature cover the pieces and resins presented in the previous page: PA, PA-GF, PC, PE, ABS, PUR, PP, PP/EPDM, PVC/ABS/PUR/PP, which represent only a fraction of the plastics present in an ELV (less than 20 kg, i.e. less than 15-20% of all plastics).

No LCA study was identified focusing on other ELV plastic components. Even if LCA data are available for some of other resins (e.g., PVC), they are not available for all of them (e.g., POM, PBT, SMA). But even if they were, they would not be sufficient since the way a resin reacts during the different end-of-life treatment alternatives can vary depending on its association with other resins.

In this study, we will thus not be able to extrapolate the conclusions we will draw for these 20 kg of plastics to all plastic components contained in ELV (more than 100 kg).

In addition, according to plastics experts, **the other resins are more difficult to recycle and thus their environmental profile is expected to be not as good (or worse) than resins assessed**.

8.1.3 General methodology developed

LCA based-approach

The purpose of this section is to give an overview of the environmental impacts related to various treatment options for plastics from ELV.

Generally speaking, the establishment of recycling and recovery targets is expected to cause both positive and negative environmental consequences. The positive consequences are associated with the control of ELV currently disposed of, but also with the use of recovered -rather than virgin- materials (which can therefore avoid the environmental impacts due to the production of virgin materials). However, in addition to these environmental benefits, environmental impacts have to be included: those caused by additional activities required to separate and recycle or recover ELV materials, including, *inter alia*, additional separation processes, transport associated with delivery to recycling/recovery facilities and the recycling/recovery processes themselves.

Thus, the control of ELV currently disposed of, the principal objective which drives the options under study, will induce a change in the balance of environmental impacts due to additional recycling and recovery activities.

Therefore, analysis and assessment have to be done through a life cycle approach. The life cycle assessment (LCA) methodology is fairly well developed and can reasonably well support comparisons of environmental benefits of various ELV disposal options. LCA is regarded by many as the most rigorous scientific approach available to quantify environmental impacts of a given 'system' (i.e. the activities to which the technique is applied).

ISO 14040 defines: *"LCA studies the environmental aspects and potential impacts throughout a product's life (i.e. cradle-to-grave) from raw material acquisition through production use and disposal. The general categories or environmental impacts needing consideration include resource use, human health and ecological consequences"*.

LCA is a decision support tool supplying information on the environmental effects of products or process. It provides information on the environmental effects and potential impacts of all the stages of product / process life cycle (from “cradle to grave”), by:

- Compiling an inventory of relevant inputs and outputs of a system throughout its entire lifecycle
- Assessing the potential environmental impacts associated with those inputs and outputs
- Interpreting the results of the inventory analysis and impact assessment phases in relation to the objectives of the study

The methodology of LCA is still under development, but a great part of standardisation has been achieved. Standards in the ISO 14040 series describe principles and framework and the four stages of an LCA:

- Step 1 - Goal definition and scope (ISO 14040 and 14041)The products/processes to be assessed are defined, a functional basis for comparison is chosen and the required level of detail is described.
- Step 2 - Inventory analysis (ISO 14041) The inputs - energy and raw materials used - and outputs - emissions to the atmosphere, water and land - are quantified for each process and then combined an inventory table (life cycle inventory, LCI).

  In an inventory table, there is a row for each substance (called an ‘elementary flow’ such as water consumption, CO_2 emissions, etc.) and a quantity in each column corresponding to a specific step of the life cycle under analysis. It is common to have up to 300 rows in an inventory table.

- Step 3 - Impact assessment (ISO 14042) The effects of the resources used and emissions generated are grouped and quantified into a limited number of impact categories which may then be weighted for importance.
- Step 4 – Interpretation / improvement assessment (ISO 14043) The results are reported in the most informative way as possible and the need and opportunities to reduce the impact of the product(s) on the environment are systematically evaluated.

Within the resources available for the study, we focused on available peer-reviewed LCAs. **We did not generate new primary data (inputs and outputs) but instead spend considerable amount of time working with available data to answer the specific questions under consideration here**. We directly considered the environmental impacts assessed in available literature (we did not re-calculate them from the inventories of inputs and outputs).

Environmental impacts quantified

The inventory table is the most objective result of a LCA study. However, a list of substances is difficult to interpret. To make this task easier, life cycle impact assessment (LCIA) is used to evaluate the environmental impacts.

From data available in published LCA studies compiled during our work, it was possible to quantify the following impacts.

Table 8.3: Environmental impact categories quantified in this study

Energy consumption	MJ
Greenhouse effect (direct, 100 yrs)	g CO_2 eq.
Air acidification	g SO_2 eq.
Photochemical oxidation	g ethylene eq.
Water pollution	critical volume in m3
Eutrophication	g PO_4 eq.
Municipal waste	kg
Hazardous waste	kg

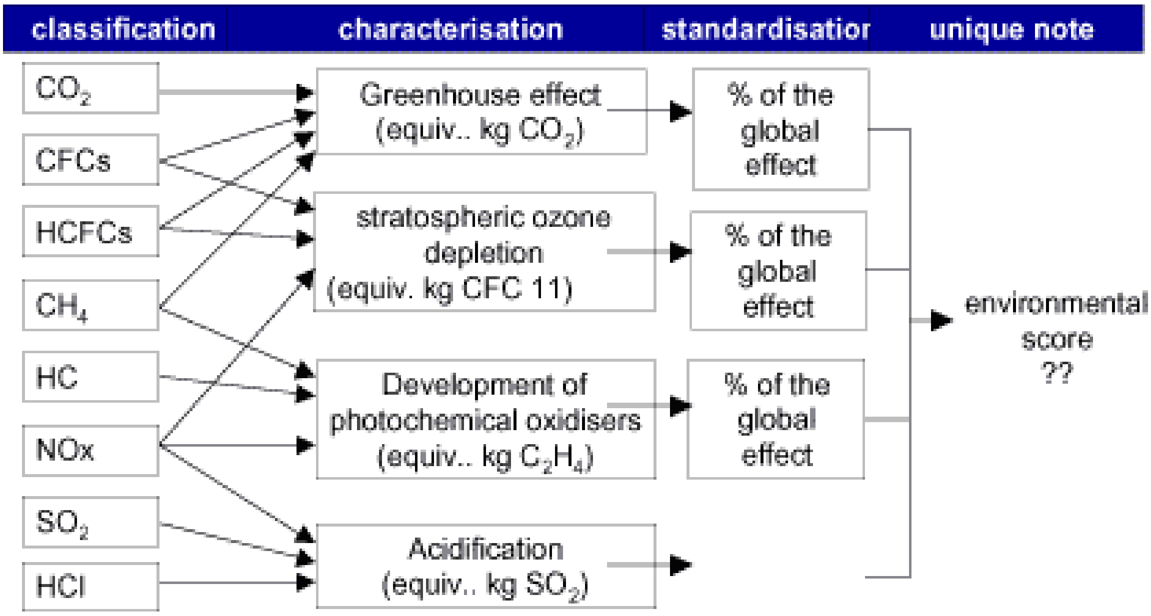
Each of them is presented in Appendix 5, as well as associated characterisation factors.

Two other impact categories were approached qualitatively:

- Land use: Land use does not only have an impact on a certain surface. The space around this area is also affected. The evaluation of the environmental impacts caused by land use is based on a relation between surface and number of species in this area. The loss of biodiversity depends on the type of grounds that are exploited as well as the surface of the area and the duration of the exploitation.
- Non renewable resources depletion: Environmental impact linked to the use of raw materials should be viewed primarily in terms of the depletion of scarce environmental resources. A total impact potential of all raw materials assessed, in the sense of assessing potential impact in terms of a single equivalence value, does not appear feasible, since the environmental impacts connected with consumption of different raw materials cannot be compared with one another.

A general approach to calculate environmental impacts from the elementary flows quantified in the LC inventory step is described hereafter with consistency to ISO standards related to LCA (ISO 14042, 14043).

Figure 8.4: Calculation of environmental impacts from the elementary flows



Classification step: all substances are sorted into classes according to the effect they have on the environment. For example, substances that contribute to the greenhouse effect or that contribute to ozone layer depletion are divided into two classes. Certain substances are included in more than one class. For example, NOx is found to be toxic, acidifying and causing eutrophication.

Characterisation step: the substances are aggregated within each class to produce an effect score. It is not sufficient just to add up the quantities of substances involved without applying weightings. Some substances may have a more intense effect than others. This problem is dealt with by applying weighting factors (so called characterisation factors) to the different substances.

Table 8.5: Example of characterisation step for a small inventory table

Emissions are multiplied by the corresponding weighting factor before being summed per class. The results are the effect scores.

Emission	Quantity (kg)	Greenhouse	Ozone layer depletion	Human toxicity	Acidification
CO2	1.792	**x 1**	-	-	-
CO	0.000670	-	-	**x 0.012**	-
NOx	0.001091	-	-	**x 0.78**	**x 0.7**
SO2	0.000987	-	-	**x 1.2**	**x 1**
Effect scores:		1.792	0	0.00204	0.0017

The interpretation of these scores may be less confusing than interpretation of a substance list, but is by no means without problems. If all the scores for one product are higher than those for another, it is easy enough to conclude which is the more environmentally friendly. But if one has a higher score for acidification, while the other has a higher score for the greenhouse effect, it becomes difficult to justify such a conclusion.

Interpretation depends on two factors:

- The relative size of the effect compared to the size of the other effects. In this example, it is important to see whether the ecotoxicity score of 100% refers to a very high or an extremely low effect level. This is normalisation.
- The relative importance attached to the various environmental effects. This is evaluation.

Normalisation (or standardisation): in order to gain a better understanding of the relative size of an effect, a normalisation step is required. Each effect calculated for the life cycle of a product is benchmarked against the known total effect for this class. However, this step is still debatable and this study does not propose any standardisation approach.

Evaluation of the normalised effect scores: normalisation considerably improves our insight into the results. However, no final judgment can be made as not all effects are considered to be of equal importance. In the evaluation phase the normalized effect scores are multiplied by a weighting factor representing the relative importance of the effect. However, this step requires accurate, complicated and… debatable system constructions; therefore, this study does not propose any unique note in order to aggregate heterogeneous environmental scores.

Remark: it should be reminded that LCAs assess **potential impacts and not actual impacts**[60]. The term 'potential' covers three characteristics of LCAs:

- The assessment of LC environmental impacts is dependent on the current **scientific knowledge** and existing models, which is intrinsically **limited**.
- Environmental impacts are assessed and aggregated from **inputs and outputs** occurring at different life cycle stages which means **with different space and time location**.

  When the environmental impact studied is global (e.g. global warming) and the inputs or outputs are cumulative (e.g. greenhouse gases), this does not make any difference.

  But this is when the environmental impact is local (e.g. air acidification) or the inputs / outputs are not cumulative (e.g. noise) that the aggregation of inputs / outputs contribution to the studied environmental impact results in potential impacts. For instance, adding up local impacts as noise and odour does not make a lot of sense because they are not global and cumulative impacts but rather dependent on the location of the "emissions".

  Thus LCAs assess **maximum potential environmental impacts** as if all the inputs and outputs occur at a same location in space and time.
- For a given physical phenomenon (e.g. air acidity), LCAs do not quantify "endpoint" impacts (such as in monetarisation methods: respiratory diseases caused by an increase of air acidity, etc.); rarely "midpoint" impacts (e.g. photochemical ozone creation potential) but generally **"start point" impacts,** i.e. the influence that pollutants emitted can have on the state of the environment (air acidity in that example). It gives a scale to assess the contribution to the environmental impact but not a quantification of the environmental impact itself (the higher the impact value quantified in LCA, the higher the environmental impact, without quantifying it directly).

Table 8.6: Start, Mid and End Point Environmental Impacts - E.g. for air acidification

Type of impact	***Scope***	***Unit***	***Where it is quantified***
Start point impact	Quantity of air emissions which influence air acidity	g SO2 equivalent	LCAs
Mid point impact	Air acidification (i.e. increase of air acidity) due to pollutants emitted	Proton concentration in the air (acidity quantity) g H^+ / m^3	Impact studies
End point impact	Social impacts of air acidification on human and ecosystems (such as respiratory diseases)	e.g. number of years of life lost	External cost analyses

'Differential systems'

An end-of-life management system treats waste while producing in the meantime material (recycling) and/or energy (energy recovery). Without the recovery of waste as material or energy, the latter would need to be produced from natural resources taken directly from the

60 This specificity of LCA addressing potential and not actual impacts concerns only the environment impacts assessment step. This does not concern the LCI step where inputs and outputs are quantified for each stage individually. It is only when one adds the different step that the "potentiality" issue occurs.

environment and transformed by industrial processes that could have more important impacts than the recovery processes.

Thus, the application of LCA consists in taking into account:

- The **generated impacts**: those generated by treatment and recycling sites, by the production of the material and the energy consumed by the sites, by transport, etc.
- The **avoided impacts** thanks to material recycling (spared raw material), heat and electricity production during waste treatment (spared fossil resources), production of methanol during waste treatment (spared fossil resources), etc.

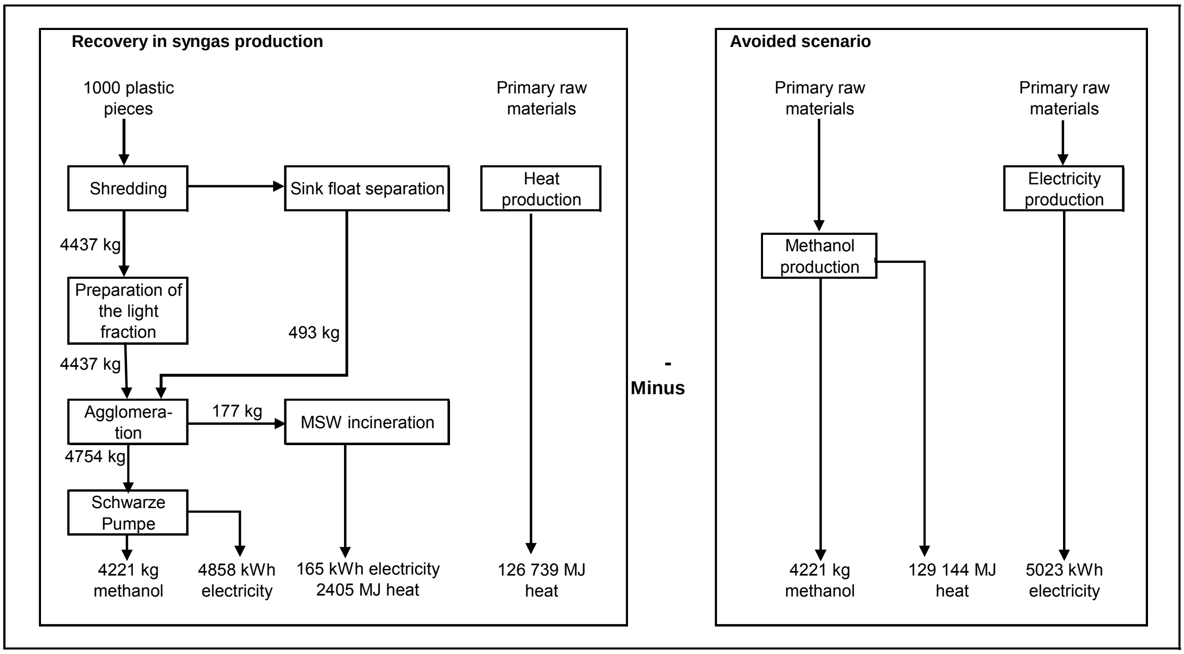
As a result, the environmental profile of a waste treatment option can either be favourable or harmful to the environment:

- It is **favourable to the environment** (i.e. with negative values corresponding to net avoided impacts) when the avoided impacts are higher than the generated impacts
- It is **harmful to the environment** (positive value, net generated impacts) when the generated impacts are greater than the avoided impacts

Taking into account the avoided impacts is necessary for the comparison of an end-of-life option with material or energy recovery (recycling, blast furnace, syngas production, cement kiln, waste incineration for instance) with another option simply disposing of the waste (landfill).

As shown in the example below, the system boundaries of the avoided scenario are determined by the boundaries of the recovery scenario studied: both scenarios must produce the same elements in order to compare. And another way to say what precedes: if the impacts of the recovery scenario which correspond to the generated impacts are more important than the impacts of the avoided scenario, then the overall impacts will have a positive value and correspond to a prejudice to the environment. On the contrary, if the total impacts have a negative value, this means that the avoided impacts are greater than the generated impacts and this is thus beneficial to the environment.

Figure 8.1: Example for the recovery of bumper in syngas production



How to read this figure: the recovery with syngas production of 1000 plastic bumper pieces from ELVs produces 4,221 kg of methanol, 2,405 MJ of heat, and 5,023 kWh of electricity. If the plastic bumper pieces had not been recovered, this methanol, heat, and electricity would have been produced from raw materials. Thus, the syngas production scenario receives a credit for the substitution of primary raw material (natural gas, waste oil and brown coal) for methanol production and fossil resources for electricity production.

Remark: in the specific case of methanol production described here, a co-product is produced: heat (129,144 MJ). In order to be comparable to the avoided scenario, the recovery scenario has thus to integrate the production of 126,739 MJ of heat from raw materials (in addition to the 2,405 MJ recovered from plastics).

The system analysed for each treatment option is presented in appendix 6.

Data sources

An extensive literature review was performed allowing identifying close to 20 studies dealing with environmental impacts of ELV end-of-life options (see appendix 8). Two LCA studies focusing on plastics contained in ELV were selected:

- "Verwertung von Kunststoffbauteilen aus Altautos – Analyse des Umwelteffekte nach dem LCA – Prinzip und ökonomische Analyse", Fraunhofer Institut für Verfahrenstechnik und Verpackung (Till Nürrenbach, Dr. Gertraud Goldhan, Alexandra Woköck), May 2002 (in the rest of this report, we will refer to it as **'Fraunhofer, 2002'**)
- « Recovery options for plastic parts from end-of-life vehicles: an eco-efficiency assessment » for APME, by Öko-Institut e.V., May 2003 (**'APME, 2003'**)

Both of them are peer-reviewed studies; this implies that the methodology followed is conformed to the ISO 14040 series and that data and assumptions are transparent enough for LCA experts to judge on the quality of the results.

In the next sections (one per resin or mix of resins), the relative positioning (in terms of environmental impacts and benefits) of the different end of life options are described. The results represent the difference between two systems (recovery systems minus avoided systems) for each of the considered impact categories (greenhouse effect, air acidification, etc.).

For a defined impact category, when the result represents the difference between two important values that are very close respectively for the recovery system and the avoided system, the relative positioning of the recovery option greatly depends on the reliability of the data and possible uncertainties. This is due to the fact that if a difference of 1% occurs in either the recovery system or the avoided system, the result of the difference between both systems can either double (change significantly), or see its tendency inverted (from positive -generated impact- the result can become negative -avoided impact- and vice-versa). Thus when comparing different end-of-life scenarios, it is important to take this into account. The same goes when a difference of 10% occurs either on the avoided or generated impacts. This shows that little is necessary for improvements to be made.

Detailed data per material/piece are attached in appendix 6.

Note: In the following, the **substitution rate** (SR) is the quantity of virgin material (in kg) that can be substituted by 1 kg of recyclates in the end product in order to achieve an equivalent performance. For example, if a 1 kg plastic part made from virgin material could only be substituted by 1 kg of recyclates, then SR=1, whereas if a 500 g plastic part made from virgin material could only be substituted by 1 kg of recyclates, then SR=0.5.

Functional unit

1 kg of each resin.

External costs

Externalities (or external costs) are the costs imposed on society and the environment that are not accounted for by the producers and consumers, i.e. which are not included in market prices. They include damage to the natural and built environment, such as effects of air pollution on health, buildings, crops, forests and global warming; occupational disease and accidents; and reduced amenity from visual intrusion of plant or emissions of noise.

The integration of a financial axis in LCA allows policy makers to get a picture of the approximate financial implications of environmental impacts linked to product or process life cycles. The 'IPP study' performed by BIO for the Commission in 2003[61] was a first attempt in developing a suitable methodology to integrate external costs in LCAs.

In the present study, the purpose was not to elaborate a new methodology. Instead we started from what was developed in this 'IPP study': we used the external costs factors compiled and applied them to the environmental impacts (and benefits) quantified here.

61 *'IPP study' = Study on External Environmental Effects Related to the Life Cycle of Products and Services, by BIO Intelligence Service for European Commission - DG Env, February 2003 (page 71) http://europa.eu.int/comm/environment/ipp/pdf/ext_effects_finalreport.pdf*

Method used for environmental impacts monetisation

For each environmental impact, the calculation method consists in:

$$EI \times ECFei = ECei$$

Where

EI = quantification of the environmental impact under consideration (e.g. for air acidification, X g SO_2 equivalent)

ECFei = external cost factor related to the environmental impact EI under consideration (e.g. for air acidification, Y Euros / g SO_2 equivalent)

ECei = external cost obtained for the environmental impact (in Euros)

The total external cost EC is then the sum of the ECei of all the environmental impacts assessed.

Table 8.7: External cost factors used

		MIN	MAX
Air acidification (g SO_2 eq.)	Euros/g	1,46E-04	1,46E-03
Greenhouse effect (direct, 100 yrs) (g CO_2 eq.)	Euros/g	1,90E-05	4,80E-05
Photochemical oxidation (g ethylene eq.)	Euros/g	7,30E-04	9,30E-04
Eutrophication (g PO_4 eq.)	Euros/g	1,54E-03	1,54E-03
Disamenity [62] (kg of waste)	Euros/kg	4,00E-03	1,90E-02

Source: Various sources compiled by BIO IS, 2003[63] *(incl. ExternE, CML, Spadaro & Rabi)*

Remark: Ranges are used for external cost factors to reflect the diversity of values existing in literature for the environmental impacts monetised.

Limitations: apart from the uncertainties which are directly linked to the monetisation methods themselves, some limits occur when combining results from monetisation and LCA.

One limit of the overall approach is linked to the fact that it combines potential global impacts (LCA) with actual location and source-specific external cost factors (monetisation).

On one hand, the environmental impacts quantified through an LCA approach are indeed both potential and global:

- Potential because the actual fate of the impact factors (emissions) in the environment and the exposure of natural systems (humans and other living systems) to these impact factors are not considered in the computational models used in LCA approach.
- Global because emissions which occur in different locations at different times are simply summed throughout a product system lifecycle. This method is valid for emissions which contribute to an environmental impact in a cumulative manner (greenhouse gases or ozone depleting substances). But for others impact categories (human health, ecotoxicology, eutrophication…), this method conducts to an overstatement of actual effects.

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62 Disamenity caused by waste incineration or landfilling: local nuisance impacts including odour, noise, dust, litter….

63 'IPP study' by BIO IS (page 71)

On the other hand, monetisation methods aim to address the location and source-specific nature of impacts associated with emissions to air, water, land. For instance, the implications of emissions from a 50 m stack are very different to those at ground level.

These general limits are further discussed in the 'IPP study' performed by BIO in section 1.3.3.3.

Another limitation comes from the fact that the scope of external costs and the scope of environmental impacts quantified in LCA do not coincide based on the current state of the art:

- External cost factors do not exist for all the environmental impacts quantified in LCA. In the present study, available external cost factors cover air acidification, climate change, photochemical oxidation, eutrophication and disamenity linked to waste but not energy consumption and water pollution.
- On the contrary, some environmental impacts not quantified in LCAs may generate external costs. In the present study, stratospheric ozone depletion and human toxicity for example are not quantified (however external cost factors would be available).

As a consequence, the external costs calculated in this study are likely to be underestimated or overestimated depending on the level of impacts or benefits without being able to quantify the gap. But because the external costs linked to greenhouse effect are preponderant[64], **external costs presented in the rest of this report can be considered giving useful orders of magnitude.**

8.1.4 Results

This section is composed of 3 parts:

- First, detailed figures per treatment options and per resin are presented, highlighting the environmental benefits on the one hand, and on the environmental disbenefits on the other hand.
- The different recovery and recycling options are then compared to landfill per resin and per treatment option. For an easier understanding, the results are presented qualitatively.
- Each recovery treatment option is eventually compared to mechanical recycling resin per resin. Here too the results are presented qualitatively to facilitate the reading.

8.1.4.1 Detailed figures per treatment option and per resin

In this section the environmental impacts and benefits obtained for the resins analysed in the Fraunhofer LCA study (PP/EPDM, PA-GF, PUR, and PVC/ABS/PP-TV/PUR) and in the APME study (PE, PC, PA, PP, PUR, ABS) and detailed in appendix 6 are summarised:

- The first table covers all resins and cases analysed (different substitution rates, different spared resources…).
- The second table qualitatively shows in which cases there are environmental benefits or disbenefits per resin and treatment option.
- The third table focuses on the cases resulting in environmental benefits (negative values, avoided impacts).
- The fourth table focuses on the cases resulting in environmental disbenefits (positive values, generated impacts).

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64 Greenhouse effect explains more than 80% of total external costs assessed for all options except landfill and 35-40% for landfill (the biggest proportion of external costs for landfill coming from disamenity due to municipal waste) - see annex 6

Table 8.8: Ranges of impacts per treatment option – All plastic resins (per kg)

Note: this table summarises results obtained with respect to the following resins: PP/EPDM, PA-GF, PUR, PVC/ABS/PP-TV/PUR, PE, PC, PA, PP, PUR, and ABS

Broad Treatment Option	Mechanical recycling		Feedstock recovery				Energy recovery				Landfill	
Detailed Treatment Option			Blast furnace		Syngas production		Cement kiln		MSWI			
Range	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Energy consumption (MJ)	-105,2	12,7	-47,67	-19,9	-58,2	-17	-48	-18,6	-35	-12,7	0,2	0,62
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	-6090	3983	-293	110	-163	1420	-1670	-588	301	2131	32,6	364
Air acidification (g SO2 eq.)	-45,6	3,1	-3,19	0,5	-11,4	2,7	-0,9	0,8	-4,1	0,33	0,01	1,5
Photochemical oxidation (*10-1 g ethylene eq.)	-358	98	-6,9	1	-54,2	3	-1,4	8,3	-4,4	2,8	0	1,4
Water pollution (critical volume in liter)	-1075	-10,8	-0,7	17,5	-77	37,7	-6	4,7	-100,1	8,74	0,6	47,44
Eutrophication (*10-2 g PO4 eq.)	-530	75	-14	11	-102	34	-3	14	-29	26	3	85
Municipal waste (g)	-272	70	-10	30	-150	12	-390	0	-70	230	1000	1001
Hazardous waste (g)	-30	11	0,1	10	-0,1	3	0	0	0	50	0	0
External costs (Euros)	-1,58E-01	2,08E-01	-6,79E-03	7,03E-03	-1,09E-02	7,32E-02	-3,36E-02	-2,61E-02	4,07E-03	1,09E-01	4,67E-03	4,01E-02

The ranges presented in the table above cover negative and positive values which are interpreted as benefits or disbenefits for the environment respectively. The results are now presented per resin in the three tables below separating the negative values which correspond to avoided impacts (and thus environmental benefits), and positive values corresponding to generated impacts (and thus environmental disbenefits). For a quick general overview, the next table qualitatively shows with a colour code in which cases there are environmental benefits (GREEN) or disbenefits (ORANGE) per resin and treatment option (LIGHT BLUE corresponds to cases for which no clear conclusion can be drawn because it depends on key parameters (substitution rates, spared resources…).

Table 8.9: Environmental benefits or disbenefits per resin and treatment option (green corresponds to environmental benefits, orange to environmental disbenefits, light blue to cases for which no clear conclusion can be drawn because it depends on key parameters (substitution rates, spared resources…)

Mechanical recycling

	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	green	light blue	green	green	green	green	green	green	green	green
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	green	light blue	green	green	orange	green	green	green	green	green
Air acidification (g SO2 eq.)	green	light blue	green	green	green	green	green	green	green	green
Photochemical oxidation (*10-1 g ethylene eq.)	green	light blue	green	green	green	green	green	green	green	green
Water pollution (critical volume in liter)	green	light blue	green	green	green	green	green	green	green	green
Eutrophication (*10-2 g PO4 eq.)	green	light blue	green	green	green	green	green	green	green	green
Municipal waste (g)	green	green	green	green	green	green	green	orange	green	green
Hazardous waste (g)	green	green	green	green	orange	green	green	green	green	green
External costs (Euros)	green	light blue	green	green	orange	green	green	green	green	green

Landfill

	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange
Air acidification (g SO2 eq.)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange
Photochemical oxidation (*10-1 g ethylene eq.)	orange	orange	orange	orange	orange	orange	orange	green	green	orange
Water pollution (critical volume in liter)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange
Eutrophication (*10-2 g PO4 eq.)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange
Municipal waste (g)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange
Hazardous waste (g)	green	green	green	green	green					
External costs (Euros)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange

Feedstock recovery – Blast furnace

	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	green	green	green	green	green	green	green	green	green	green
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	green	orange	green	orange	orange	green	orange	orange	green	light blue
Air acidification (g SO2 eq.)	light blue	orange	green	orange	orange	green	green	green	green	green
Photochemical oxidation (*10-1 g ethylene eq.)	light blue	green	green	green	green	green	green	green	green	green
Water pollution (critical volume in liter)	orange	orange	green	orange	orange	green	orange	orange	green	light blue
Eutrophication (*10-2 g PO4 eq.)	orange	orange	green	orange	orange	green	orange	orange	green	green
Municipal waste (g)	green	green	green	orange	orange	green	orange	orange	green	green
Hazardous waste (g)	orange	orange	green	orange	orange	green	green	green	green	light blue
External costs (Euros)	green	orange	green	orange	orange	green	orange	orange	green	green

Feedstock recovery – Syngas production

	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	green	green	green	green	green	green	green	green	green	green
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	light blue	orange	orange	orange	orange	green	orange	orange	orange	green
Air acidification (g SO2 eq.)	light blue	orange	green	light blue	orange	green	green	green	green	green
Photochemical oxidation (*10-1 g ethylene eq.)	light blue	orange	green	light blue	orange	green	green	green	green	green
Water pollution (critical volume in liter)	light blue	green	orange	light blue	green	orange	orange	orange	orange	orange
Eutrophication (*10-2 g PO4 eq.)	light blue	orange	green	light blue	orange	green	green	green	green	green
Municipal waste (g)	light blue	orange	green	light blue	orange	green	green	green	green	light blue
Hazardous waste (g)	light blue	orange	green	orange	orange	green	green	green	green	green
External costs (Euros)	light blue	orange	orange	orange	orange	green	green	orange	green	green

Energy recovery – Cement kiln

	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	green	green	green	green	green	green	green	green	green	green
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	green	green	green	green	green	green	green	orange	green	green
Air acidification (g SO2 eq.)	green	green	orange	green	orange	orange	orange	orange	orange	orange
Photochemical oxidation (*10-1 g ethylene eq.)	green	green	orange	green	green	orange	orange	orange	orange	light blue
Water pollution (critical volume in liter)	light blue	orange	orange	orange	orange	orange	orange	orange	orange	orange
Eutrophication (*10-2 g PO4 eq.)	light blue	orange	orange	orange	orange	orange	orange	orange	orange	orange
Municipal waste (g)	green	green	green	green	green	green	green	green	green	green
Hazardous waste (g)	green	green	green	green	green	green	green	green	green	green
External costs (Euros)	green	green	green	green	green	green	green	orange	green	green

Energy recovery – MSWI

	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	green	green	green	green	green	green	green	green	green	green
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange
Air acidification (g SO2 eq.)	green	green	orange	green	green	orange	green	orange	orange	orange
Photochemical oxidation (*10-1 g ethylene eq.)	green	green	green	green	green	green	orange	green	green	green
Water pollution (critical volume in liter)	green	green	orange	green	green	orange	orange	orange	orange	orange
Eutrophication (*10-2 g PO4 eq.)	green	orange	orange	orange	orange	orange	orange	orange	orange	orange
Municipal waste (g)	green	green	green	orange	orange	green	orange	orange	green	green
Hazardous waste (g)	orange	orange	green	orange	orange	green	orange	orange	green	light blue
External costs (Euros)	orange	orange	orange	orange	orange	orange	orange	orange	orange	orange

Table 8.10: Cases for which there are environmental benefits (negative value; avoided impact) per resin and treatment option (per kg)

Note: orange cells correspond to environmental disbenefits; those values are presented in the table following this one

Mechanical recycling

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	-105,2 to -20	-57	-95 for SR=1	-67,83	-105,2	-29 to -20	-67,33	-94,21	-77,77	-74,3	-65,85 to -50,35
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	-6090 to -934,74	-992	-3638 for SR=1	-2042,5	-6090		-1046,51	-5838,89	-2956,7	-1740,7	-1231,53 to -934,74
Air acidification (g SO2 eq.)	-45,6 to -2,39	-17,1	-25 for SR=1	-15,83	-45,6	-6,5 to -4,7	-15,98	-2,39	-16,03	-11,63	-15,54 to -12,21
Photochemical oxidation (*10-1 g ethylene eq.)	-358,3 to -2	-7,2	-12 for SR=1	-9,6	-23,9	-2,3 to -2	-21,4	-358,3	-16	-14,4	-8,4 to -6,4
Water pollution (critical volume in liter)	-1075 to -10,8	-18	-406 to -100	-1075	-343	-10,9 to -10,8	-31,63	-416,39	-362,43	-120,74	-30,96 to -23,37
Eutrophication (*10-2 g PO4 eq.)	-530 to -1	-78	-380 for SR=1	-237	-530	-3 to -1	-91	-463	-162	-113	-91 to -71
Municipal waste (g)	-272 to 0	-20	-254 to -77	-30	-243	-272 to -268	0	-10		0	0
Hazardous waste (g)	-30 to 0	-8	-29,9 to -7,5	-30	-13,6		0	-10	0	0	0
External costs (Euros)											

Feedstock recovery – Blast furnace

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	-47,67 to -19,9	-44 to -37	-29,7	-30,49	-25,8	-27	-47,67	-21	-19,9	-38,85	-47,39 to -26,61
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	-293 to -32	-293 to -32		-88,33			-167,44			-55,56	-165,61 for bumper
Air acidification (g SO2 eq.)	-3,19 to -0,07	-0,3 when S=heavy oil		-1,92			-3,19	-0,07	-0,53	-2,33	-3,15 to -1,16
Photochemical oxidation (*10-1 g ethylene eq.)	-6,9 to -0,7	-3,5 when S=heavy oil	-2,3	-1,3	-2	-2,1	-2,6	-6,9	-0,7	-1,9	-2,5 to -1,1
Water pollution (critical volume in liter)				0			-0,7			-0,37	-0,64 for bumper
Eutrophication (*10-2 g PO4 eq.)				-4			-14			-8	-14 to 0
Municipal waste (g)	-1 to 0	-1 to -0,4	-0,2	-10			0			0	0
Hazardous waste (g)	0			0			0	0	0	0	0 for bumper

Feedstock recovery – Syngas production

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	-58,19 to -17	-49 to -17	-22,9	-35,18	-25,8 to -19,9	-21	-58,19	-27,96	-26,8	-48,22	-57,85 to -34,35
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	-162,8 to -74,74	-160 when S=methanol from WO					-162,8				-161,46 to -74,74
Air acidification (g SO2 eq.)	-11,42 to -0,42	-3,4 when S=methanol from WO		-2,5	-1,7 when S=methanol from waste oil		-11,42	-0,42	-5,17	-9,44	-11,34 to -6,63
Photochemical oxidation (*10-1 g ethylene eq.)	-54,2 to -0,7	-1,5 when S=methanol from WO		-0,7	-0,7 when S=methanol from waste oil		-8,6	-54,2	-4	-7	-8,6 to -5,2
Water pollution (critical volume in liter)	-7,7 to -2,5	-7,7 to -2,5	-3,6		-2,5 when S=methanol from waste oil	-3,49					
Eutrophication (*10-2 g PO4 eq.)	-102 to -10	-23 when S=methanol from WO		-13	-10 when S=methanol from waste oil		-102	-48	-46	-84	-101 to -59
Municipal waste (g)	- 90 to -15	-29 when S=methanol from WO		-150	-15 when S=methanol from waste oil		-90	-30	-30	-70	-90 for bumper
Hazardous waste (g)	-0,1 to 0	-0,1 when S=methanol from WO		0			0	0	0	0	0

Energy recovery – Cement kiln

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	-48 to -18,63	-48 to -44	-26,7	-25,62	-25,5	-26	-42,81	-19,57	-18,63	-35,11	-42,56 to -24,47
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	-1670 to -369,44	-1670 to -1488	-578	-310	-747	-734	-1104,65	-369,44		-500	-1099,04 to -588,42
Air acidification (g SO2 eq.)	- 0,9 to -0,03	- 0,9 to -0,2	-0,03		-0,1						
Photochemical oxidation (*10-1 g ethylene eq.)	-1,4 to 0	-1,4 to 0	-0,8		-0,8	-0,8					-0,01 for bumper
Water pollution (critical volume in liter)	-0,6	-0,6 when S=brown coal									
Eutrophication (*10-2 g PO4 eq.)	-3	-3 when S=brown coal									
Municipal waste (g)	-390 to 0	-32 to -0,2	-0,1	-370	-0,1	-0,1	-50	-390	0	-40	-50 to 0
Hazardous waste (g)	0	0	0	0	0	0	0	0	0	0	0

Energy recovery – MSWI

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	-35 to -12,7	-35 to -17	-14,5	-19,49	-12,7	-14	-31,67	-15,17	-14,5	-26,26	-31,51 to -18,63
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)											
Air acidification (g SO2 eq.)	-4,1 to -0,06	-4,1 to -0,9	-0,5		-0,4	-0,3		-0,06			
Photochemical oxidation (*10-1 g ethylene eq.)	-4,4 to -0,03	-4,4 to -0,3	-1,3	-0,7	-1,2	-1,3	-0,12		-0,03	-0,11	-0,11 to -0,06
Water pollution (critical volume in liter)	-10,1 to -0,9	-10,1 to -0,9	-3,7		-3,5	-3,32					
Eutrophication (*10-2 g PO4 eq.)	-29 to -5	-29 to -5									
Municipal waste (g)	-70 to 0	-32 to -0,5	-9	-40			-70			-40	-70 to 0
Hazardous waste (g)	0			0			0			0	0 for bumper

Table8.11: Cases for which there are environmental disbenefits (positive value; generated impact) per resin and treatment option (per kg)

Note: green cells correspond to environmental benefits; those values are presented in the previous table

Landfill

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	0,2 to 0,53	0,2	0,2	0,62	0,2	0,2	0,53	0,53	0,5	0,52	0,52 to 0,53
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	32,56 to 364	364	269	36,67	237	248	32,56	33,33	33,33	33,33	32,63 to 32,80
Air acidification (g SO2 eq.)	0,01 to 1,5	0,1	0,7	0,25	1,5	0,4	0,21	0,01	0,2	0,22	0,19 to 0,21
Photochemical oxidation (*10-1 g ethylene eq.)	0,1 to 1,4	1,1	0,8	0,1	0,7	0,8	0,2	1,4			0,1
Water pollution (critical volume in liter)	0,6 to 47,44	0,6	0,6	35,83	0,6	0,6	47,44	46,94	47,2	47,04	47,20 to 47,26
Eutrophication (*10-2 g PO4 eq.)	2 to 85	2	38	3	85	18	18	18	18	18	18
Municipal waste (g)	1000 to 1001	1001	1001	1000	1001	1001	1000	1000	1000	1000	1000
Hazardous waste (g)											

Mechanical recycling

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)	12,7		12,7 for SR=0,65								
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	395 to 3983		3983 for SR=0,65			395 to 595					
Air acidification (g SO2 eq.)	3,1		3,1 for SR=0,65								
Photochemical oxidation (*10-1 g ethylene eq.)	98		98 for SR=0,65								
Water pollution (critical volume in liter)											
Eutrophication (*10-2 g PO4 eq.)	75		75 for SR=0,65								
Municipal waste (g)	70								70		
Hazardous waste (g)	2 to 11					2 to 11					

Feedstock recovery

Blast furnace

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)											
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	9,47 to 110		71		92	85		63,89	110		9,47 for air duct
Air acidification (g SO2 eq.)	0,1 to 0,5	0,5 when S=hard coal	0,1		0,2	0,14					
Photochemical oxidation (*10-1 g ethylene eq.)	1	1 when S=hard coal									
Water pollution (critical volume in liter)	0,74 to 1,75	15 to 17	17		17	17,5		0,83	0,97		0,74 for air duct
Eutrophication (*10-2 g PO4 eq.)	4 to 11	6 to 11	9		9	9		4	5		
Municipal waste (g)	2 to 30				2	2,4		30	30		
Hazardous waste (g)	0,1 to 10	0,1	0,1		0,6	1					10 for air duct

Syngas production

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)											
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	29,17 to 1420	1297 to 1420 when S=methanol from mix	1168	660,83	225 to 1045	1047		29,17	380	262,96	
Air acidification (g SO2 eq.)	1,3 to 2,7	2,1 to 2,7 when S=methanol from mix	1,4		1,4 when S=methanol from mix	1,3					
Photochemical oxidation (*10-1 g ethylene eq.)	1,7 to 3,0	3 when S=methanol from mix	1,8		1,8 when S=methanol from mix	1,7					
Water pollution (critical volume in liter)	17 to 37,67			22,5	17 when S=methanol from mix		37,67	19,58	18,87	31,48	24 to 37,55
Eutrophication (*10-2 g PO4 eq.)	10 to 34	12 to 34 when S=methanol from mix	11		10 when S=methanol from mix	10					
Municipal waste (g)	6 to 10	11 to 12 when S=methanol from mix	6		8 when S=methanol from mix	8,6					10 for air duct
Hazardous waste (g)	0,04 to 3	0,1 to 3 when S=methanol from mix	0,04		0,5 to 0,6	1					

Energy recovery

Cement kiln

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)											
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	3,33								3,33		
Air acidification (g SO2 eq.)	0,04 to 0,8			0,67		0,05	0,23	0,04	0,8	0,41	0,22 to 0,63
Photochemical oxidation (*10-1 g ethylene eq.)	0,3 to 8,3			0,3				8,3	0,3		0,3 for air duct
Water pollution (critical volume in liter)	1,11 to 5	5 when S=hard coal	5	1,67	5	4,7	1,16	1,53	1,57	1,11	1,18 to 1,58
Eutrophication (*10-2 g PO4 eq.)	3 to 14	3 when S=hard coal	4	12	4	4	9	14	14	10	9 to 13
Municipal waste (g)											
Hazardous waste (g)											

MSWI

	General	PP/EPDM	PUR (Fraunhofer)	PUR (APME)	PA-6,6 GF	12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV	PE	PA	PC	ABS	PP
Energy consumption (MJ)											
Greenhouse effect (direct, 100 yrs) (g CO2 eq.)	301 to 2131	301 to 2131	1146	1282,5	909	971	1540	840,28	1157	1674,1	916,34 to 1531,21
Air acidification (g SO2 eq.)	0,21 to 0,33			0,25			0,33		0,27	0,33	0,21 to 0,32
Photochemical oxidation (*10-1 g ethylene eq.)	2,8							2,8			
Water pollution (critical volume in liter)	6,77 to 8,74			5,83			8,14	6,94	6,77	7,04	7,99 to 8,74
Eutrophication (*10-2 g PO4 eq.)	8 to 26		8	17	8	8	26	14	15	23	17 to 26
Municipal waste (g)	44 to 230				44	63,3		220	230		
Hazardous waste (g)	1,9 to 50	3	1,9		16,7	32		40	30		50 for air duct

Remark: **Differences can be seen** not only between different types of resins but also for a same resin **from one study to another**. For example, PUR in blast furnace is beneficial for all impact categories in the APME study whereas it is beneficial in terms of energy consumption, photochemical oxidation and municipal waste and harmful in terms of greenhouse effect, air acidification, water pollution, eutrophication, and hazardous waste in the Fraunhofer study. These differences in the results between the two studies could be explained by differences in the following:

- The system studied
- Geographical representativeness of data
- Data quality
- Choice of substitution (spared resource…)

Comparing in such detail both studies requires further investigations which are extremely time-consuming; it was not the purpose of the present project. However, to facilitate access to information and comprehension (the original report is available in German only), the results of the Fraunhofer report are compiled and commented in great details in appendix 6.

8.1.4.2 Landfill versus recovery

Hereafter, the results from Fraunhofer and APME studies are qualitatively summarised as landfill compared to the recovery options. An 'R' indicates lower impacts for the recovery option considered (recovery is better for the impact category considered) and an 'L' indicates lower impacts for landfill (landfill is better for the impact category considered). 'R=L' indicates that there is no significant difference between landfill and the recovery option. No consideration has been given to the magnitude of the difference. Data are nevertheless detailed in appendix 6. The following results are independent from the quantity of plastic collected from ELVs.

The results are successively given for each type of resin studied.

As previously mentioned (see §8.1.2), five types of end-of-life recovery options were considered: mechanical recycling, blast furnace, syngas production, cement kiln, and waste incineration.

PP | ***Landfill versus Recovery***

Main conclusions

- For all the impact categories under study, mechanical recycling has a better environmental profile than landfill.
- Furthermore, the mechanical recycling option leads to significant benefits with respect to the 8 impact categories under study (see table of values in appendix 6: all the values are negative)
- The environmental profile of the other recovery options is more contrasted
- The comparison of each other recovery option with the landfill scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category
- Nevertheless certain recovery options (syngas production, cement kiln, and blast furnace according to the APME study) can have a better profile than landfill for all impact categories depending on the raw material (waste oil in the syngas production scenario) or the type of fuel (brown coal in cement kiln) spared thanks to the recovery of PP/EPDM

Detailed data and detailed analysis

See appendix 6

Summary of key results

Resin	**PP/EPDM**	**PP (air duct)**	**PP (bumper)**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery	mechanical recycling	mechanical recycling	mechanical recycling
	post-dismantling	post-dismantling	post-dismantling
Technology	PP	PP (air duct)	PP (bumper)
Substitution	SR = 1	SR = 1	SR = 1
Representativeness	site specific	specific data	specific data
Geography	Germany	Western Europe	
Landfill	landfill		
Technology	post-shredding		
Representativeness	average	site specific	
Geography	Germany	Western Europe	
Energy consumption	R	R	R
Global warming potential	R	R	R
Air acidification potential	R	R	R
Photochemical oxidation	R	R	R
Water pollution	R	R	R
Eutrophication	R	R	R
Municipal waste	R	R	R
Hazardous waste	R		

SR = substitution rate
nat. gas = natural gas
WO = waste oil

R: recovery option is better than landfill
L: landfill is better than the recovery option
R=L: no signicant difference between landfill and the recovery option

Resin	**PP/EPDM**	**PP/EPDM**	**PP (air duct)**	**PP (bumper)**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	blast furnace	blast furnace	blast furnace	blast furnace
Substitution	heavy oil	hard coal	heavy oil	heavy oil
Representativeness	pilot scale	pilot scale	average	average
Geography	Germany		Western Europe	
Landfill	landfill			
Technology	post-shredding			
Representativeness	average		site specific	
Geography	Germany		Western Europe	
Energy consumption	R	R	R	R
Global warming potential	R	R	R	R
Air acidification potential	R or R=L (?)	L or R=L (?)	R	R
Photochemical oxidation	R	R	R	R
Water pollution	L	L	R	R
Eutrophication	L	L	R	R
Municipal waste	R	R	R	R
Hazardous waste	R=L	R=L		

Resin	**PP/EPDM**	**PP/EPDM**	**PP/EPDM**	**PP (air duct)**	**PP (bumper)**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling
Technology	post-shredding	post-dismantling	post-shredding	post-shredding	post-shredding
	syngas production	syngas production	syngas production	syngas production	syngas production
Substitution	73,4% nat. gas, 22,1% WO, 4,5% brown coal	73,4% nat. gas, 22,1% WO, 4,5% brown coal	WO	?	?
Representativeness	site specific	site specific	site specific	site specific	site specific
Geography	Germany			Germany	
Landfill	landfill				
Technology	post-shredding				
Representativeness	average			site specific	
Geography	Germany			Western Europe	
Energy consumption	R	R	R	R	R
Global warming potential	L	L	R	R	R
Air acidification potential	L	L	R	R	R
Photochemical oxidation	L	L	R	R	R
Water pollution	R	R	R	R	R
Eutrophication	L	L	R	R	R
Municipal waste	R	R	R	R	R
Hazardous waste	R=L	L	R=L		

Resin	**PP/EPDM**	**PP/EPDM**	**PP (air duct)**	**PP (bumper)**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	cement kiln	cement kiln	cement kiln	cement kiln
Substitution	hard coal	brown coal	48% coal and 52% lignite	48% coal and 52% lignite
Representativeness	pilot scale	pilot scale	average	average
Geography	Germany		Western Europe	
Landfill	landfill			
Technology	post-shredding			
Representativeness	average		site specific	
Geography	Germany		Western Europe	
Energy consumption	R	R	R	R
Global warming potential	R	R	R	R
Air acidification potential	R=L or ?	R	L	L
Photochemical oxidation	R	R	L	R
Water pollution	L	R	R	R
Eutrophication	R=L	R	R	R
Municipal waste	R	R	R	R
Hazardous waste	R=L	R=L		

Resin	**PP/EPDM**	**PP/EPDM**	**PP/EPDM**	**PP (air duct)**	**PP (bumper)**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery	energy recovery	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
	MSWI	MSWI	MSWI	MSWI	MSWI
Substitution	electricity + steam	electricity	steam	electricity + steam	electricity + steam
Representativeness	average	average	average	average	average
Geography	Germany			Western Europe	
Landfill	landfill				
Technology	post-shredding				
Representativeness	average			site specific	
Geography	Germany			Western Europe	
Energy consumption	R	R	R	R	R
Global warming potential	L	L	R=L	L	L
Air acidification potential	R	R	R	R=L	L
Photochemical oxidation	R	R	R	R	R
Water pollution	R	R	R	R	R
Eutrophication	R	R	R	R	L
Municipal waste	R	R	R	R	R
Hazardous waste	L	L	L		

PUR from seat cushion — ***Landfill versus Recovery***

Main conclusions

- The environmental profile of the recovery options is contrasted
- In general, **the comparison of each other recovery option with the landfill scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category**
- Nevertheless certain recovery options (mechanical recycling, blast furnace) can have a better profile than landfill for all impact categories depending on the substitution rate for mechanical recycling and except for one impact category, water pollution, for the blast furnace option in the Fraunhofer study
- Furthermore, when the substitution rate is 1, the mechanical recycling option leads to significant benefits with respect to the 8 impact categories under study (see table of values in appendix 6: all the values are negative)

Detailed data and detailed analysis

See appendix 6

Summary of key results

Resin	**PUR**	**PUR**	**PUR**	**PUR**	**PUR**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	APME (2003)
Recovery	mechanical recycling	mechanical recycling	mechanical recycling	feedstock recycling	feedstock recycling
Technology	post-dismantling	post-dismantling	post-dismantling	post-shredding	post-shredding
	PUR	PUR	PUR	blast furnace	blast furnace
Substitution	SR = 1	SR = 0,65	SR = 1	heavy oil	heavy oil
Representativeness	site specific	site specific	specific data	pilot scale	average
Geography	Germany		Western Europe	Germany	Western Europe
Landfill	landfill				
Technology	post-shredding				
Representativeness	average		site specific	average	site specific
Geography	Germany		Western Europe	Germany	Western Europe
Energy consumption	R	L	R	R	R
Global warming potential	R	L	R	R	R
Air acidification potential	R	L or R=L (?)	R	R=L	R
Photochemical oxidation	R	L	R	R	R
Water pollution	R	R	R	L	R
Eutrophication	R	L	R	R	R
Municipal waste	R	R	R	R	R
Hazardous waste	R	R	R	R=L	R=L

SR = substitution rate
nat. gas = natural gas
WO = waste oil

R: recovery option is better than landfill
L: landfill is better than the recovery option
R=L: no signicant difference between landfill and the recovery option

Resin	**PUR**	**PUR**	**PUR**	**PUR**	**PUR**	**PUR**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	APME (2003)
Recovery	feedstock recycling	feedstock recycling	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
	syngas prod.	syngas production	cement kiln	cement kiln	MSWI	MSWI
Substitution	73,4% nat gas, 22,1% WO, 4,5% brown coal	?	hard coal	48% coal and 52% lignite	electricity + steam	electricity + steam
Representativeness	site specific	site specific	pilot scale		average	national
Geography	Germany	Germany	Germany	Western Europe	Germany	Western Europe
Landfill	landfill					
Technology	post-shredding					
Representativeness	average	site specific	average	site specific	average	site specific
Geography	Germany	Western Europe	Germany	Western Europe	Germany	Western Europe
Energy consumption	R	R	R	R	R	R
Global warming potential	L	L	R	R	L	L
Air acidification potential	L	R	R=L	L	R=L (?)	R=L
Photochemical oxidation	L	R	R	L	R	R
Water pollution	R	R	L	R	R	R
Eutrophication	R	R	R	L	R	L
Municipal waste	R	R	R	R	R	R
Hazardous waste	R=L	R=L	R=L	R=L	L	R=L

PA-6.6 GF from hubcap

Landfill versus Recovery

Main conclusions

- For all the impact categories under study, mechanical recycling has a better environmental profile than landfill
- Furthermore, the mechanical recycling option leads to significant benefits with respect to the 8 impact categories under study (see table of values in appendix 6: all the values are negative)
- The environmental profile of the other recovery options is more contrasted
- The comparison of each other recovery option with the landfill scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category
- Nevertheless certain recovery options (syngas production) can have a better profile than landfill for all impact categories or for all impact categories but one (photochemical oxidation) depending on the raw material (waste oil in the syngas production scenario) spared thanks to the recovery of PA

Detailed data and detailed analysis

See appendix 6

Summary of key results

Resin	**PA-6,6 GF**	**PA**	**PA-6,6 GF**	**PA**	**PA-6,6 GF**	**PA-6,6 GF**	**PA**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)
Recovery	mechanical recycling	mechanical recycling	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling
Technology	post-dismantling	post-dismantling	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
	recycling of PA	PA	blast furnace	blast furnace	syngas production	syngas production	syngas production
Substitution	SR = 1	SR = 1	heavy oil	heavy oil	73,4% nat gas, 22,1% WO, 4,5% brown coal	WO	?
Representativeness	site specific	specific data	pilot scale	average	site specific	site specific	site specific
Geography	Germany	Western Europe	Germany	Western Europe	Germany	Germany	Germany
Landfill	landfill						
Technology	post-shredding						
Representativeness	average	site specific	average	site specific	average	average	site specific
Geography	Germany	Western Europe	Germany	Western Europe	Germany	Germany	Western Europe
Energy consumption	R	R	R	R	R	R	R
Global warming potential	R	R	R	L	L	R	R
Air acidification potential	R	R	R or L=R?	R	R=L	R	R
Photochemical oxidation	R	R	R	R	L	R	R
Water pollution	R	R	L	R	L	R	R
Eutrophication	R	R	R	R	R	R	R
Municipal waste	R	R	R	R	R	R	R
Hazardous waste	R		R=L		R=L	R=L	

SR = substitution rate
nat. gas = natural gas
WO = waste oil

R: recovery option is better than landfill
L: landfill is better than the recovery option
R=L: no signicant difference between landfill and the recovery option

Resin	**PA-6,6 GF**	**PA**	**PA-6,6 GF**	**PA**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	APME (2003)
Recovery	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	cement kiln	cement kiln	MSWI	MSWI
Substitution	hard coal	48% coal and 52% lignite	electricity + steam	electricity + steam
Representativeness	pilot scale	average	average	average
Geography	Germany	Western Europe	Germany	Western Europe
Landfill	landfill			
Technology	post-shredding			
Representativeness	average	site specific	average	site specific
Geography	Germany	Western Europe	Germany	Western Europe
Energy consumption	R	R	R	R
Global warming potential	R	R	L	L
Air acidification potential	R or L=R?	L	R	R
Photochemical oxidation	R	L	R	L
Water pollution	L	R	R	R
Eutrophication	R	R	R	R
Municipal waste	R	R	R	R
Hazardous waste	R=L		L	

12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV from dashboard

Landfill versus Recovery

Main conclusions

- Certain recovery options (cement kiln, blast furnace, mechanical recycling with the recycling of PP and PVC) have a better environmental profile than landfill for all impact categories except for one: water pollution for cement kiln and blast furnace, global warming potential for mechanical recycling
- The environmental profile of the recovery options is contrasted
- The comparison of each other recovery option with the landfill scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category

Detailed data and detailed analysis

See appendix 6

Summary of key results

Resin	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)
Recovery	mechanical recycling	mechanical recycling	feedstock recycling	feedstock recycling	energy recovery	energy recovery
	post-dismantling	post-dismantling	post-shredding	post-shredding	post-shredding	post-shredding
Technology	recycling of the PP-beam	recycling of the PP and PVC&recycling of the particles plate in a similar particle panel	blast furnace	syngas production	cement kiln	MSWI
Substitution	SR = 1 for the PP-beam	SR = 1 for PP and PVC	heavy oil	73,4% nat gas, 22,1% WO, 4,5% brown coal	hard coal	elec + steam
Representativeness	pilot scale	pilot scale	pilot scale	site specific	pilot scale	average
Landfill	landfill					
Technology	post-shredding					
Representativeness	average					
Geography	Germany	Germany	Germany	Germany	Germany	Germany
Energy consumption	R	R	R	R	R	R
Global warming potential	L	L	R	L	R	L
Air acidification potential	R	R	R	L	R or R=L (?)	R
Photochemical oxidation	R	R	R	L	R	R
Water pollution	R	R	L	R	L	R
Eutrophication	R	R	R	R	R	R
Municipal waste	R	R	R	R	R	R
Hazardous waste	L	L or R=L (?)	L or R=L (?)	L or R=L (?)	R=L	L

SR = substitution rate
nat. gas = natural gas
WO = waste oil

R: recovery option is better than landfill
L: landfill is better than the recovery option
R=L: no signicant difference between landfill and the recovery option

PE from wash tank and lid

Landfill versus Recovery

Main conclusions

- Certain recovery options (blast furnace, mechanical recycling and syngas production) have a better environmental profile than landfill for all impact categories
- One recovery option (cement kiln) has a better environmental profile than landfill for all impact categories except for one: air acidification potential
- The environmental profile of the recovery options is contrasted
- The comparison of the waste incineration recovery option with the landfill scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category

Detailed data

Refer to APME report directly.

Summary of key results

Resin	**PE**	**PE**	**PE**	**PE**	**PE**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	APME (2003)	APME (2003)	APME (2003)	APME (2003)	APME (2003)
Recovery	mechanical recycling	feedstock recycling	feedstock recycling	energy recovery	energy recovery
	post-dismantling	post-shredding	post-shredding	post-shredding	post-shredding
Technology	PE	blast furnace	syngas production	cement kiln	MSWI
Substitution	SR = 1	heavy oil	?	48% coal and 52% lignite	electricity + steam
Representativeness	specific data	average	site specific	average	average
Geography	Western Europe	Western Europe	Germany	Western Europe	Western Europe
Landfill	landfill				
Technology	post-shredding				
Representativeness	site specific				
Geography	Western Europe				
Energy consumption	R	R	R	R	R
Global warming potential	R	R	R	R	L
Air acidification potential	R	R	R	L	L
Photochemical oxidation	R	R	R	R	R
Water pollution	R	R	R	R	R
Eutrophication	R	R	R	R	L
Municipal waste	R	R	R	R	R
Hazardous waste					

SR = substitution rate

R: recovery option is better than landfill

L: landfill is better than the recovery option

R=L: no signicant difference between landfill and the recovery option

PC from headlamp lens | ***Landfill versus Recovery***

Main conclusions

- Mechanical recycling has a better environmental profile than landfill for all impact categories
- Certain recovery option (blast furnace, and syngas production) have a better environmental profile than landfill for all impact categories except for one: global warming potential
- The environmental profile of the recovery options is contrasted
- **The comparison of the cement kiln and the waste incineration recovery option with the landfill scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category**

Detailed data

Refer to APME report directly.

Summary of key results

Resin	**PC**	**PC**	**PC**	**PC**	**PC**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	APME (2003)	APME (2003)	APME (2003)	APME (2003)	APME (2003)
Recovery	mechanical recycling	feedstock recycling	feedstock recycling	energy recovery	energy recovery
	post-dismantling	post-shredding	post-shredding	post-shredding	post-shredding
Technology	PC	blast furnace	syngas production	cement kiln	MSWI
Substitution	SR = 1	heavy oil	?	48% coal and 52% lignite	electricity + steam
Representativeness	specific data	average	site specific	average	average
Geography	Western Europe	Western Europe	Germany	Western Europe	Western Europe
Landfill	landfill				
Technology	post-shredding				
Representativeness	average				
Geography	Western Europe				
Energy consumption	R	R	R	R	R
Global warming potential	R	L	L	R	L
Air acidification potential	R	R	R	L	L
Photochemical oxidation	R	R	R	L?	R
Water pollution	R	R	R	R	R
Eutrophication	R	R	R	R	R
Municipal waste	R	R	R	R	R
Hazardous waste					

SR = substitution rate
R: recovery option is better than landfill
L: landfill is better than the recovery option
R=L: no signicant difference between landfill and the recovery option

ABS from mirror housing

Landfill versus Recovery

Main conclusions

- Mechanical recycling and blast furnace have a better environmental profile than landfill for all impact categories
- Certain recovery option (cement kiln, and syngas production) have a better environmental profile than landfill for all impact categories except for one: air acidification potential for cement kiln, and global warming potential for syngas production
- The environmental profile of the recovery options is contrasted
- The comparison of the waste incineration recovery option with the landfill scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category

Detailed data

Refer to APME report directly.

Summary of key results

Resin	**ABS**	**ABS**	**ABS**	**ABS**	**ABS**
Scenarios compared	**R vs L**	**R vs L**	**R vs L**	**R vs L**	**R vs L**
Source	APME (2003)	APME (2003)	APME (2003)	APME (2003)	APME (2003)
Recovery	mechanical recycling	feedstock recycling	feedstock recycling	energy recovery	energy recovery
	post-dismantling	post-shredding	post-shredding	post-shredding	post-shredding
Technology	recycling of ABS	blast furnace	syngas production	cement kiln	MSWI
Substitution	SR = 1	heavy oil	?	48% coal and 52% lignite	electricity + steam
Representativeness	specific data	average	site specific Germany	average	average
Geography	Western Europe	Western Europe	Germany	Western Europe	Western Europe
Landfill	landfill				
Technology	post-shredding				
Representativeness	average				
Geography	Western Europe				
Energy consumption	R	R	R	R	R
Global warming potential	R	R	L	R	L
Air acidification potential	R	R	R	L	L
Photochemical oxidation	R	R	R	R=L	R
Water pollution	R	R	R	R	R
Eutrophication	R	R	R	R	L
Municipal waste	R	R	R	R	R
Hazardous waste					

SR = substitution rate
R: recovery option is better than landfill
L: landfill is better than the recovery option
R=L: no signicant difference between landfill and the recovery option
Western Europe=EU 15, Switzerland and Norway

8.1.4.3 *Mechanical recycling versus other Recovery options*

Hereafter, the results from Fraunhofer and APME studies are qualitatively summarised as mechanical recycling compared to the other recovery options. An 'MR' indicates lower impacts for mechanical recycling (mechanical recycling is better for the impact category considered) and an 'R' indicates lower impacts for the other recovery option considered (recovery is better for the impact category considered). 'R=MR' indicates that there is no significant difference between mechanical recycling and the other recovery option. No consideration has been given to the magnitude of the difference. Data are nevertheless detailed in appendix 6. The following results are independent from the quantity of plastic collected from ELVs.

The results are given for each type of resin studied. As in the previous section, here too there are some differences between the available studies.

PP	Mechanical recycling vs other Recovery options

Main conclusions

- For all the impact categories under study, mechanical recycling has a better environmental profile than blast furnace
- The environmental profile of the other recovery options is more varied when compared to mechanical recycling
- For certain recovery options, the comparison of each other recovery option with the mechanical recycling scenario is to the advantage of the mechanical recycling option except for one impact category: water pollution in the case of waste incineration, and global warming in the case of cement kiln

Detailed data and detailed analysis

See appendix 6

Summary of key results

Resin	**PP/EPDM**	**PP/EPDM**	**PP (air duct)**	**PP (bumper)**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery (R)	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	blast furnace	blast furnace	blast furnace	blast furnace
Substitution	heavy oil	hard coal	heavy oil	heavy oil
Representativeness	pilot scale	pilot scale	average	average
Geography	Germany		Western Europe	
Recovery (MR)	mechanical recycling			
Technology	post-dismantling			
	recycling of PP			
Substitution	SR=1			
Representativeness	site specific		specific data	
Geography	Germany		Western Europe	
Energy consumption	MR	MR	MR	MR
Global warming potential	MR	MR	MR	MR
Air acidification potential	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR
Water pollution	MR	MR	MR	MR
Eutrophication	MR	MR	MR	MR
Municipal waste	MR	MR	MR=R	MR=R
Hazardous waste	MR	MR	MR	MR=R

SR = substitution rate
nat. gas = natural gas
WO = waste oil

R: recovery option is better than mechanical recycling
MR: mechanical recycling is better than the recovery option

Resin	**PP/EPDM**	**PP/EPDM**	**PP/EPDM**	**PP (air duct)**	**PP (bumper)**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery (R)	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling
Technology	post-shredding	post-dismantling	post-shredding	post-shredding	post-shredding
	syngas production	syngas production	syngas production	syngas production	syngas production
Substitution	73,4% nat. gas, 22,1% WO, 4,5% brown coal	73,4% nat gas, 22,1% WO, 4,5% brown coal	waste oil	?	?
Representativeness	site specific	site specific	site specific	site specific	site specific
Geography	Germany			Germany	
Recovery (MR)	mechanical recycling				
Technology	post-dismantling				
	recycling of PP				
Substitution	SR=1				
Representativeness	site specific			specific data	
Geography	Germany			Western Europe	
Energy consumption	MR	MR	MR	MR	MR
Global warming potential	MR	MR	MR	MR	MR
Air acidification potential	MR	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR	R?
Water pollution	R	R	R	MR	MR
Eutrophication	MR	MR	MR	MR	R
Municipal waste	MR	MR	R	MR	R
Hazardous waste	MR	MR	MR	MR=R	MR=R

Resin	PP/EPDM	PP/EPDM	PP (air duct)	PP (bumper)
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery (R)	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	cement kiln	cement kiln	cement kiln	cement kiln
Substitution	hard coal	brown coal	48% coal and 52% lignite	48% coal and 52% lignite
Representativeness	pilot scale	pilot scale	average	average
Geography	Germany		Western Europe	
Recovery (MR)	mechanical recycling			
Technology	post-dismantling			
	recycling of PP			
Substitution	SR=1			
Representativeness	site specific		specific data	
Geography	Germany		Western Europe	
Energy consumption	MR	MR	MR	MR
Global warming potential	R	R	MR	MR
Air acidification potential	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR
Water pollution	MR	MR	MR	MR
Eutrophication	MR	MR	MR	MR
Municipal waste	MR	R	MR=R	R
Hazardous waste	MR	MR	MR=R	MR=R

Resin	PP/EPDM	PP/EPDM	PP/EPDM	PP (air duct)	PP (bumper)
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	APME (2003)
Recovery (R)	energy recovery	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
	MSWI	MSWI	MSWI	MSWI	MSWI
Substitution	electricity + steam	electricity	steam	electricity + steam	electricity + steam
Representativeness	average	average	average	average	average
Geography	Germany			Western Europe	
Recovery (MR)	mechanical recycling				
Technology	post-dismantling				
	recycling of PP				
Substitution	SR=1				
Representativeness	site specific			specific data	
Geography	Germany			Western Europe	
Energy consumption	MR	MR	MR	MR	MR
Global warming potential	MR	MR	MR	MR	MR
Air acidification potential	MR	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR	MR
Water pollution	R	R	MR	MR	MR
Eutrophication	MR	MR	MR	MR	MR
Municipal waste	MR	MR	R	MR=R	R
Hazardous waste	MR	MR	MR	MR	MR=R

PUR from seat cushion

Mechanical recycling vs other Recovery options

Main conclusions

- The comparison of each other recovery option with the mechanical recycling scenario greatly depends on the substitution factor in the mechanical recycling scenario.
- When the substitution rate is 1, for all the impact categories under study, mechanical recycling has a better environmental profile than the other recovery options except for one impact category in the APME study: municipal waste.
- When the substitution rate is 0.65, the comparison of each other recovery option with the mechanical recycling scenario is either to the advantage of the recovery option, or to the advantage of the landfill scenario depending on the considered impact category. Concerning the impacts on resources and air, the other recovery options are better than mechanical recycling. However, the mechanical recycling scenario comes out better than the other recovery options for most of the impacts on water and waste.

Detailed data and detailed analysis

See appendix 6

Summary of key results

Resin	**PUR**	**PUR**	**PUR**	**PUR**	**PUR**	**PUR**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)
Recovery (R)	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling
Technology	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
	blast furnace	blast furnace	blast furnace	syngas prod.	syngas prod.	syngas production
Substitution	heavy oil	heavy oil	heavy oil	73,4% nat gas, 22,1% WO, 4,5% brown coal	73,4% nat gas, 22,1% WO, 4,5% brown coal	?
Representativeness	pilot scale	pilot scale	average	site specific	site specific	site specific
Geography	Germany		Western Europe	Germany		Germany
Recovery (MR)	mechanical recycling					
Technology	post-dismantling					
	recycling of PUR					
Substitution	SR=1	SR=0,65	SR=1	SR=1	SR=0,65	SR=1
Representativeness	site specific		specific data	site specific		specific data
Geography	Germany		Western Europe	Germany		Western Europe
Energy consumption	MR	R	MR	MR	R	MR
Global warming potential	MR	R	MR	MR	R	MR
Air acidification potential	MR	R	MR	MR	R	MR
Photochemical oxidation	MR	R	MR	MR	R	MR
Water pollution	MR	MR	MR	MR	MR	MR
Eutrophication	MR	R	MR	MR	R	MR
Municipal waste	MR	MR	MR	MR	MR	R
Hazardous waste	MR	MR	MR	MR	MR	MR

SR = substitution rate
nat. gas = natural gas
WO = waste oil

R: recovery option is better than mechanical recycling
MR: mechanical recycling is better than the recovery option

Resin	**PUR**	**PUR**	**PUR**	**PUR**	**PUR**	**PUR**
Scenarios com	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)
Recovery (R)	energy recovery	energy recovery	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
	cement kiln	cement kiln	cement kiln	MSWI	MSWI	MSWI
Substitution	hard coal	hard coal	48% coal and 52% lignite	electricity + steam	electricity + steam	electricity + steam
Representativeness	pilot scale	pilot scale	average	average	average	average
Geography	Germany		Western Europe	Germany		Western Europe
Recovery (MR)	mechanical recycling					
Technology	post-dismantling					
	recycling of PUR					
Substitution	SR=1	SR=0,65	SR=1	SR=1	SR=0,65	SR=1
Representativeness	site specific		specific data	site specific		specific data
Geography	Germany		Western Europe	Germany		Western Europe
Energy consump	MR	R	MR	MR	R	MR
Global warming	MR	R	MR	MR	R	MR
Air acidification	MR	R	MR	MR	R	MR
Photochemical o	MR	R	MR	MR	R	MR
Water pollution	MR	MR	MR	MR	MR	MR
Eutrophication	MR	R	MR	MR	R	MR
Municipal waste	MR	MR	R	MR	MR	R
Hazardous wast	MR	MR	MR	MR	MR	MR

PA-6.6 GF from hubcap

Mechanical recycling vs other Recovery options

Main conclusions

For all the impact categories under study, **mechanical recycling has a better environmental profile than all the other recovery options**; except for one impact category for the cement kiln and syngas production options: municipal waste. Furthermore, the mechanical recycling option leads to significant benefits with respect to the 8 impact categories under study (see table of values in appendix 6: all the values are negative).

Detailed data and detailed analysis

See appendix 6

Summary of key results

Resin	**PA-6,6 GF**	**PA**	**PA-6,6 GF**	**PA-6,6 GF**	**PA**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	Fraunhofer (2002)	APME (2003)
Recovery (R)	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling
Technology	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
	blast furnace	blast furnace	syngas production	syngas production	syngas production
Substitution	heavy oil	heavy oil	73,4% nat gas, 22,1% WO, 4,5% brown coal	WO	?
Representativeness	pilot scale	average	site specific	site specific	site specific
Geography	Germany	Western Europe	Germany	Germany	Germany
Recovery (MR)	mechanical recycling				
	post-dismantling				
Technology	recycling of PA-6,6 GF	recycling of PA	recycling of PA-6,6 GF	recycling of PA-6,6 GF	recycling of PA
Substitution	SR=1				
Representativeness	average	specific data	average	average	specific data
Geography	Germany	Western Europe	Germany	Germany	Western Europe
Energy consumption	MR	MR	MR	MR	MR
Global warming potential	MR	MR	MR	MR	MR
Air acidification potential	MR	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR	MR
Water pollution	MR	MR	MR	MR	MR
Eutrophication	MR	MR	MR	MR	MR
Municipal waste	MR	MR	MR	MR	R
Hazardous waste	MR	MR	MR	MR	MR

SR = substitution rate
nat. gas = natural gas
WO = waste oil

MR: mechanical recycling is better than the recovery option

Resin	**PA-6,6 GF**	**PA**	**PA-6,6 GF**	**PA**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	APME (2003)	Fraunhofer (2002)	APME (2003)
Recovery (R)	energy recovery	energy recovery	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	cement kiln	cement kiln	MSWI	MSWI
Substitution	hard coal	coal and lignite	electricity + steam	electricity + steam
Representativeness	pilot scale	average	average	average
Geography	Germany	Western Europe	Germany	Western Europe
Recovery (MR)	mechanical recycling			
Technology	post-dismantling			
	recycling of PA-6,6 GF	recycling of PA	recycling of PA-6,6 GF	recycling of PA
Substitution	SR=1			
Representativeness	average	specific data	average	specific data
Geography	Germany	Western Europe	Germany	Western Europe
Energy consumption	MR	MR	MR	MR
Global warming potential	MR	MR	MR	MR
Air acidification potential	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR
Water pollution	MR	MR	MR	MR
Eutrophication	MR	MR	MR	MR
Municipal waste	MR	R	MR	MR
Hazardous waste	MR	MR	MR	MR

12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV from dashboard	***Mechanical recycling vs other Recovery options***

Main conclusions

- The environmental profile of the recovery options is varied when compared to mechanical recycling
- **The comparison of each recovery option with the mechanical recycling scenario is either to the advantage of the recovery option, or to the advantage of the mechanical recycling scenario depending on the considered impact category**
- Nevertheless **the comparison of the waste incineration scenario with the mechanical recycling scenario is to the advantage of the mechanical recycling option except for one impact category**: water pollution

Detailed data and detailed analysis

See appendix 6

Summary of key results

Results are summarised in the table next page.

Resin	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**	**12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)	Fraunhofer (2002)
Recovery (R)	feedstock recycling	feedstock recycling	feedstock recycling	feedstock recycling	energy recovery	energy recovery	energy recovery	energy recovery
	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding	post-shredding
Technology	blast furnace	blast furnace	syngas production	syngas production	cement kiln	cement kiln	MSWI	MSWI
Substitution	heavy oil	heavy oil	73,4% nat gas, 22,1% WO, 4,5% brown coal	73,4% nat gas, 22,1% WO, 4,5% brown coal	hard coal	hard coal	elec + steam	elec + steam
Representativeness	pilot scale	pilot scale	site specific	site specific	pilot scale	pilot scale	average	average
Recovery (MR)	mechanical recycling							
	post-dismantling							
Technology	recycling of the PP-beam	recycling of the PP and PVC&recycling of the particles plate in a similar particle panel	recycling of the PP-beam	recycling of the PP and PVC&recycling of the particles plate in a similar particle panel	recycling of the PP-beam	recycling of the PP and PVC&recycling of the particles plate in a similar particle panel	recycling of the PP-beam	recycling of the PP and PVC&recycling of the particles plate in a similar particle panel
Substitution	SR=1							
Representativeness	pilot scale							
Geography	Germany	Germany	Germany	Germany	Germany	Germany	Germany	Germany
Energy consumption	R or MR=R?	MR or MR=R?	R	MR	R	MR	MR	MR
Global warming potential	R	R	MR	MR	R	R	MR	MR
Air acidification potential	MR	MR	MR	MR	MR	MR	MR	MR
Photochemical oxidation	R or MR=R?	MR or MR=R?	MR	MR	MR	MR	MR	MR
Water pollution	MR	MR	R	R	MR	MR	R	R
Eutrophication	MR	MR	MR	MR	MR	MR	MR	MR
Municipal waste	MR	MR	MR	MR	MR	MR	MR	MR
Hazardous waste	R	R	R	R	R	R	MR	MR

SR = substitution rate
nat. gas = natural gas
WO = waste oil

R: recovery option is better than mechanical recycling
MR: mechanical recycling is better than the recovery option
MR=R: no signicant difference between mechanical recycling and the recovery option

PE from wash tank and lid | ***Mechanical recycling vs other Recovery options***

Main conclusions

- For all impact categories under study, mechanical recycling has a better environmental profile than blast furnace
- Mechanical recycling has a better environmental profile than certain recovery option (waste incineration) for all impact categories except for one: municipal waste
- The environmental profile of the recovery options is contrasted
- The comparison of the other recovery option with the mechanical recycling scenario is either to the advantage of the recovery option, or to the advantage of the mechanical recycling scenario depending on the considered impact category

Detailed data

Refer to APME report.

Summary of key results

Resin	**PE**	**PE**	**PE**	**PE**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	APME (2003)	APME (2003)	APME (2003)	APME (2003)
Recovery (R)	feedstock recycling	feedstock recycling	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	blast furnace	syngas production	cement kiln	MSWI
Substitution	heavy oil	?	48% coal and 52% lignite	electricity + steam
Representativeness	average	site specific	average	average
Geography	Western Europe	Germany	Western Europe	Western Europe
Recovery (MR)	mechanical recycling			
Technology	post-dismantling			
	recycling of PE			
Substitution	SR=1			
Representativeness	specific data			
Geography	Western Europe			
Energy consumption	MR	MR	MR	MR
Global warming potential	MR	MR	R	MR
Air acidification potential	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR
Water pollution	MR	MR	MR	MR
Eutrophication	MR	R	MR	MR
Municipal waste	MR=R	R	R	R
Hazardous waste	MR=R	MR=R	MR=R	MR=R

SR = substitution rate
R: recovery option is better than mechanical recycling
MR: mechanical recycling is better than the recovery option
MR=R: no signicant difference between mechanical recycling and the recovery option

PC from headlamp lens

Mechanical recycling vs other Recovery options

Main conclusions

- For all impact categories under study, mechanical recycling has a better environmental profile than waste incineration
- Mechanical recycling has a better environmental profile than the other recovery option for all impact categories except for one: municipal waste

Detailed data

Refer to APME report.

Summary of key results

Resin	**PC**	**PC**	**PC**	**PC**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	APME (2003)	APME (2003)	APME (2003)	APME (2003)
Recovery (R)	feedstock recycling	feedstock recycling	energy recovery	energy recovery
Technology	post-shredding	post-shredding	post-shredding	post-shredding
	blast furnace	syngas production	cement kiln	MSWI
Substitution	heavy oil	?	48% coal and 52% lignite	electricity + steam
Representativeness	average	site specific	average	average
Geography	Western Europe	Germany	Western Europe	Western Europe
Recovery (MR)	mechanical recycling			
Technology	post-dismantling			
	recycling of PC			
Substitution	SR=1			
Representativeness	specific data			
Geography	Western Europe			
Energy consumption	MR	MR	MR	MR
Global warming potential	MR	MR	MR	MR
Air acidification potential	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR
Water pollution	MR	MR	MR	MR
Eutrophication	MR	MR	MR	MR
Municipal waste	R	R	R	MR
Hazardous waste	MR=R	MR=R	MR=R	MR

SR = substitution rate
R: recovery option is better than mechanical recycling
MR: mechanical recycling is better than the recovery option
MR=R: no signicant difference between mechanical recycling and the recovery option

ABS from mirror housing | ***Mechanical recycling vs other Recovery options***

Main conclusions

- For all impact categories under study, mechanical recycling has a better environmental profile than blast furnace
- Mechanical recycling has a better environmental profile than the other recovery option for all impact categories except for one: municipal waste

Detailed data

Refer to APME report.

Summary of key results

Resin	**ABS**	**ABS**	**ABS**	**ABS**
Scenarios compared	**R vs MR**	**R vs MR**	**R vs MR**	**R vs MR**
Source	APME (2003)	APME (2003)	APME (2003)	APME (2003)
Recovery (R)	feedstock recycling	feedstock recycling	energy recovery	energy recovery
	post-shredding	post-shredding	post-shredding	post-shredding
Technology	blast furnace	syngas production	cement kiln	MSWI
Substitution	heavy oil	?	48% coal and 52% lignite	electricity + steam
Representativeness	average	site specific	average	average
Geography	Western Europe	Germany	Western Europe	Western Europe
Recovery (MR)	mechanical recycling			
Technology	post-dismantling			
	recycling of ABS			
Substitution	SR=1			
Representativeness	specific data			
Geography	Western Europe			
Energy consumption	MR	MR	MR	MR
Global warming potential	MR	MR	MR	MR
Air acidification potential	MR	MR	MR	MR
Photochemical oxidation	MR	MR	MR	MR
Water pollution	MR	MR	MR	MR
Eutrophication	MR	MR	MR	MR
Municipal waste	MR=R	R	R	R
Hazardous waste	MR=R	MR=R	MR=R	MR=R

SR = substitution rate
R: recovery option is better than mechanical recycling
MR: mechanical recycling is better than the recovery option
MR=R: no signicant difference between mechanical recycling and the recovery option

8.1.5 Summary of key results all resins together

Important caveats: conclusions drawn below are based on the preceding results and apply to ELV plastic components analysed in available literature: bumper (PP/EPDM), hub cap (PA-GF), seat cushion (PUR), dashboard (PVC/ABS/PP-TV/PUR), wash fluid tank & lid (PE), headlamp lens (PC), intake manifold (PA), bumper & air duct (PP) and mirror housing (ABS), which represent about 15-20% of plastics contained in an ELV in 2015. In the absence of LCA available for the other plastic pieces present in ELV, it is **not possible to extrapolate these results to all plastic contained in ELV** (see section 8.1.2 for more explanations). In addition, according to plastics experts, the other plastic pieces are more difficult to recycle and thus their environmental profile is expected to be not as good (or worse) than plastic pieces assessed.

Although results are presented per resin, without systematically mentioning the ELV piece concerned, they have to be considered valid for the concerned ELV piece. It may not be correct to extrapolate them to other pieces or products made of the same resin.

Recycling versus Landfill

When the substitution rate is 1, mechanical recycling comes out more beneficial than landfill for all impact categories and for all resins studied (PA-GF, PP/EPDM, PUR, ABS, PA, PC, PE, PP) **except for** the recycling of part or all of the **dashboard** composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV.

When the substitution rate is less than 1 (which is more likely in real industrial conditions according to plastic experts), the results are much more contrasted. No general conclusion can be drawn except that **the lower the substitution rate, the lower the environmental benefits of mechanical recycling and, under a certain level of substitution rate, benefits can even be replaced by disbenefits** which can become higher that landfill impacts (for instance for PUR, this threshold is between 0.65 and 1). This is further analysed through sensitivity analyses about substitution rates in section 8.1.6.

Remark: there is one environmental impact category which is never quantified in LCA and which is important when considering landfill: land use. Landfill is known for being detrimental to land use. In cases where recycling is less beneficial than landfill for some impact categories, it is sometimes heard that, still, it is better than landfill for land use impact. In fact, this would need to be demonstrated because all the facilities involved in the recycling system also occupy land.

Other recovery options versus landfill

Cement kiln recovery option comes out better than landfill when the spared resource is brown coal (for all impact categories and for all resins considered: PA-GF, PP/EPDM, PUR, 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV). When the spared resource is hard coal, cement kiln is worse than landfill for impacts on water (for all resins).

The blast furnace recovery option comes out better than landfill when the spared resource is heavy oil for all impact categories except impacts on water (and for all resins). When the spared resource is hard coal, there is a prejudice not only for impacts on water but also for air acidification.

The syngas production recovery option comes out better or equivalent to landfill when the spared resource is waste oil for all impact categories (and for all resins). Syngas production can be worse than landfill for impacts on climate change in case of other substitution.

The results are much more contrasted for the waste incineration recovery option. It depends on the resin considered and the impact categories considered. From data available, it is only for PP/EPDM with recovered energy enabling to save steam that the waste incineration option comes out better than landfill for all indicators. For the other resins, incineration performing often worse than landfill in GWP and hazardous wastes, but better in the other 6 categories assessed. Key parameters are also the type of energy recovered and substituted as well as the efficiency rate.

Further analysis would be necessary to assess the environmental impacts/benefits of recovery options in more contrasted situations than those analysed in the 2 LCAs.

Mechanical recycling versus other recovery options

For some resins (PA, PC, ABS), mechanical recycling (with a substitution rate of 1, as analysed in the studies) **has a better environmental profile than other recovery options**, with respect to all the impact categories under study.
For other resins (PP, PUR, plastics mix from dashboard, PE), the environmental profile of the other recovery options is more varied when compared to mechanical recycling. Results depend on the resin, the recovery option, the substitution rate (for mechanical recycling) and the impact category considered, without any possibility to release general principles.

Conclusions based on external costs

From the 'smileys table' next page, let's now summarise the results from the external costs point of view, which is a way to aggregate all the environmental impacts (with the limits mentioned in section 8.1.3: all environmental impacts quantified here were not monetised by lack of external cost factor-such as water pollution / critical volume- and other impacts which are expected to contribute to external costs are not accounted for here -such as toxicity-).

1/ External costs have a profile similar to greenhouse effect (because greenhouse effect explains more than 80% of total external costs assessed for all options except landfill and 35-40% for landfill (the biggest proportion of external costs for landfill coming from disamenity due to municipal waste) – see detailed figures in appendix 6).

2/ Compared to landfill:

- are more beneficial: mechanical recycling with SR 1, cement kiln, syngas (with waste oil as spared resource), blast furnace.
- are less beneficial: mechanical recycling with SR 0,65 (only one case studied: PUR), syngas (spared resources other than waste oil), MSWI.

Another way to say it: **diverting ELV plastics pieces from landfill is beneficial except for mechanical recycling with low SR, MSWI and syngas (depending on spared resource).**

Remark about blast furnace and syngas production:

- During our discussions with the IISI, it was pointed out that recovery of plastics in **blast furnace** faces **intrinsic limits**: first only plastic fractions with high calorific value (> 35 kcal/kg) can be accepted; secondly a maximum of 60 kg of plastics / tonne of cast iron can be used.
- During discussions with the Öko-Institut, it was pointed out that a maximum of 50 000 t of plastics / year can be used in a **syngas production** plant.

3/ Compared to other recovery options, mechanical recycling is the best solution from an environmental point of view when SR = 1 (except for cement kiln when the spared energy source is brown coal) but the worst for low SR.

Summary table

The table below summarises the results of the comparison of the environmental impacts of the end-of-life options for the different plastics resins.

A ☺ means that the recovery option comes out better than landfill (resp. mechanical recycling) in terms the impact category considered, a 😐 means that the environmental impacts are equivalent[65], and a ☹ signifies that the recovery option comes out worse than landfill (resp. mechanical recycling) in terms the impact category considered.

65 Due to first intrinsic LCA data uncertainties (usually assessed at 10% by LCA practitioners) and second the fact that these uncertainties can invert the relative positioning of 2 options, a difference between 2 treatment options of less than 10% was considered non significant thus the impacts equivalent.

Table 8.11: Qualitative Summary of Comparative Environmental Impact Assessment – All plastic resins (per kg)

Note: this table summarises results obtained with respect to the following resins: PP/EPDM, PUR, PA and a mix of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV, ABS, PE, PC, PP

Comparison	Broad Treatment Option	Detailed Treatment Option	Non renewable resource depletion	Climate Change (g eq CO2)	Energy Cons. (MJ)	Water Pollution (critical vol)	Municipal Waste (g)	Air Acidification (g eq SO2)	Photochem. Oxidation (g eq ethylene)	Eutrophication (g eq PO4)	Haz. Waste (g)	Land use	External costs (Euros)
With Landfill	Mechanical Recycling	Substitution rate (SR=1)	Despite the 2 LCA studies used do not quantify this indicator, experts are used to say that this indicator comes out favourable to the recovery option compared to landfill.	☺ [1]	☺	☺	☺	☺	☺	☺	☺ [1]	Land use is not quantified in LCA. But landfill is known for being detrimental to land use. It is theoretically possible that some recovery options are better than landfill for this impact category but this would need to be demonstrated because all the facilities involved in the recovery system also occupy	☺
		Substitution rate (SR=0.65) [2]		☹	☹	☺	☺	😐	☹	☹	☺		☹
	Feedstock Recovery	Blast Furnace (S=heavy oil)		☺ [3]	☺	(4)	☺	☺ [5] or 😐 [6]	☺	☺ [7]	😐		☺
		Blast Furnace [8] (S=hard coal)		☺	☺	☹	☺	😐	☺	☹	😐		☺
		Syngas Production (S=mix 1) [10]		☹	☺	☺ [12]	☺	☹ [12]	☹	☺ [7]	😐 [13]		☹
		Syngas Production (S=waste oil) [11]		☺ [12]	☺	☺	☺	☺	☺	☺	😐 (PA-GF) ☺ (PP/EPDM)		☺
		Syngas Production (S=mix 2) [14]		15	☺	☺	☺	☺	☺	☺	😐 [16]		☺ except PUR
	Energy Recovery	Cement Kiln (S=hard coal) [17]		☺	☺	☹	☺	😐	☺	☺ [9]	😐		☺
		Cement Kiln [18] (S=brown coal)		☺	☺	☺	☺	☺	☺	☺	😐		☺
		Cement Kiln (S=48% coal and 52% brown coal) [44]		☺	☺	☺	☺	☹	☹ [19]	☺ [20]	😐		☺
		MSWI		☹ [21]	☺	☺	☺	22	☺ [23]	☺ [24]	☹ [25]		☹ except PP/EPDM steam

Comparison	Broad Treatment Option	Detailed Treatment Option	Climate Change (g eq CO2)	Energy Cons. (MJ)	Water Pollution (critical vol)	Municipal Waste (g)	Air Acidification (g eq SO2)	Photochem. Oxidation (g eq ethylene)	Eutrophication (g eq PO4)	Haz. Waste (g)	External costs (Euros)

With Mechanical Recycling (SR=1)	Feedstock Recovery	Blast Furnace (S=heavy oil)	☹ [1]	☹ [26]	☹	☹ [27]	☹	☹ [26]	☹	☹ [28]	☹ except dashboard
		Blast Furnace (S=hard coal) [8]	☹	☹	☹	☹	☹	☹	☹	☹	☹
		Syngas Production (S=mix 1) [10]	☹	☹ [29]	(30)	☹ [31]	☹	☹	☹ [32]	☹ [33]	☹
		Syngas Production (S=waste oil) [11]	☹	☹	☹ PA-GF ☺ PP/EPDM	☹ PA-GF ☺ PP/EPDM	☹	☹	☹	☹	☹
		Syngas Production (S= mix 2) [14]	☹	☹	☹	☺	☹	☹ [35]	☹ [36]	😐 [37]	☹
	Energy Recovery	Cement Kiln (S=hard coal) [17]	☹ [34]	☹ [29]	☹	☹	☹	☹	☹	☹ [1]	☺ dashboard, PP/EPDM ☹ PUR, PA
		Cement Kiln (S=brown coal)[18]	☺	☹	☹	☺	☹	☹	☹	☹	☺
		Cement Kiln (S=48% coal and 52% brown coal)[44]	☹ [38]	☹	☹	☺ [39]	☹	☹	☹	😐 [40]	☹
		MSWI	☹	☹	☹ [41]	☹ [21, 42]	☹	☹	☹	☹ [43]	☹
With Mechanical Recycling (SR=0,65) [2]	Feedstock Recovery	Blast Furnace (S=heavy oil)	☺	☺	☹	☹	☺	☺	☺	☹	☺
		Syngas Production (S=mix 2)	☺	☺	☹	☹	☺	☺	☺	☹	☺
	Energy Recovery	Cement Kiln (S=hard coal)	☺	☺	☹	☹	☺	☺	☺	☹	☺
		MSWI	☺	☺	☹	☹	☺	☺	☺	☹	☺

(1) For all resins (PA-GF, PUR, PP/EPDM, PA, PP, PE, PC, ABS) except for dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV
(2) Results for the mechanical recycling option with a substitution rate of 0,65 are available for PUR in the Fraunhofer study.
(3) For all resins except PA and PC.
(4) ☺ for all resins in the APME study (PC, PP, ABS, PUR, PA, PE) and ☹ for all resins in the Fraunhofer study (PA-GF, PP/EPDM, PUR, and 12,5% PVC, PVC/ABS/PUR/PP-TV)
(5) PP, dashboard, PA, PE, PC, ABS, PUR in the APME study
(6) PP/EPDM, PUR in the Fraunhofer study, PA-GF
(7) For all resins (PA-GF, PUR, PVC/ABS/PUR/PP-TV, PA, PP, PE, PC, ABS) except for PP/EPDM
(8) Results for the blast furnace option with substitution=hard coal are only available for PP/EPDM.
(9) Except PP/EPDM which is 😐
(10) S=mix1= 73,4% natural gas, 22,1% waste oil, 4,5% brown coal. Results for the syngas production recovery option when the spared resource for the production of methanol is waste oil alone are available for PP/EPDM and PA.
(11) Results for the syngas production recovery option when the spared resource for the production of methanol is composed of 73,4% natural gas, 22,1% waste oil, 4,5% brown coal are available for PP/EPDM, PA-GF, PUR and PVC/ABS/PUR/PP-TV in the Fraunhofer study.
(12) For all resins (PP/EPDM, PUR, PVC/ABS/PUR/PP-TV) except for PA-GF which is 😐
(13) For all resins except for PP/EPDM when syngas production occurs after dismantling of the plastic piece.
(14) S=mix2= natural gas + electricity and nitrogen. Results available for (PC, PP, ABS, PUR, PA, PE) in the APME study.
(15) ☺ for PA, PE, and PP, and ☹ for PUR, PC, and ABS.
(16) For all resins (ABS, PE, PUR, PC) in the APME study except PA and PP.
(17) Results for the cement kiln option with substitution=hard coal are available for PP/EPDM, PA-GF, PUR and PVC/ABS/PUR/PP-TV in the Fraunhofer study.
(18) Results for the cement kiln option with substitution=brown coal are only available for PP/EPDM
(19) For all resins (PA, PP in air duct, PUR, PC) in the APME study except for ABS which is equivalent and PE and PP in bumper which are☹.
(20) For all resins (PA, PP, ABS, PC, PE) in the APME study except PUR.
(21) For all resins except for PP/EPDM when the recovered energy enables to save steam alone.
(22)☺ for all resins in the Fraunhofer study (PA-GF, PP/EPDM, PUR, and 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV), ☹ for ABS, PC, PE, PP in bumper in the APME study and 😐 for PP in air duct, PUR in the APME study.
(23)For all resins (PA-GF, PUR, PVC/ABS/PUR/PP-TV, PP/EPDM, PP, PE, PC, ABS) except PA.
(24) For all resins (PA-GF, PP/EPDM, PVC/ABS/PUR/PP-TV, PUR in the Fraunhofer study, PP in air duct, PA, PC) except for ABS, PE, PP in bumper, and PUR in the APME study which are ☹.
(25) For all resins (PA-GF, PUR in the Fraunhofer study, PVC/ABS/PUR/PP-TV, PP/EPDM, PA, PP, PE, PC, ABS) except for PUR in the APME study.
(26) For all resins (PA-GF, PUR, PP/EPDM, PA, PP, PE, PC, ABS) except dashboard in PVC/ABS/PUR/PP-TV which is 😐
(27) For all resins (PA-GF, PUR, PVC/ABS/PUR/PP-TV, PP/EPDM, PA) except PP, PE, ABS in APME study where the impacts are equivalent and PC where the impacts are ☹.

(28) For all resins (PUR, PA, PA-GF, PP in air duct, PP/EPDM) except for PC, ABS, PP in bumper, PE which are equivalent and dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV where the impacts are☹.
(29) For all resins except for dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV when only PP-TV is recycled
(30) ☺ for PP/EPDM and dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV, and ☹ for PUR and PA-GF
(31) For all resins except for PP/EPDM when the spared resource for the production of methanol is waste oil alone, PP in bumper in the APME study, PUR in the APME study, PA, PC, ABS.
(32) For all resins except PP in bumper in APME study, PE.
(33)For all resins except PP, PE, ABS and PC in APME study where the impacts are equivalent and dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV which is ☹
(34) For all resins (PA-GF, PUR, PA, PP, PE, PC, ABS) except for PP/EPDM and dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV
(35) For all resins (PA, PUR, ABS, PP in air duct, PE, PC) except for PP in bumper.
(36) For all resins (PA, PUR, ABS, PP in air duct, PC) except for PP in bumper and PE.
(37) For all resins (PUR, PP, PE, PC) except for ABS and PA which are ☹.
(38)For all resins (ABS, PP, PC, PA, PUR, PA-GF, PVC/ABS/PUR/PP-TV, PP/EPDM) except PE.
(39) For all resins (ABS, PE, PC, PA, PUR) except PP in the APME study (☹ PP in bumper, 😐 PP in air duct).
(40) For all resins (ABS, PE, PP, PC) in the APME study except for PA, PUR which are ☹.
(41) For all resins (PA-GF, PUR, PP/EPDM, PA, PP, PE, PC, ABS) except PP/EPDM when the recovered energy enables to save electricity alone or electricity and steam together and dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV
(42) For all resins (PP/EPDM, PA-GF, PC, PUR in Fraunhofer study, PVC/ABS/PUR/PP-TV, PA) except for ABS, PE, PUR in APME study, and PP in bumper in APME study which are ☺, and PP in air duct which is equivalent.
(43) For all resins (PA-GF, PUR, PVC/ABS/PUR/PP-TV, PP/EPDM, PA, PP in air duct, PC) except PE, ABS, and PP in bumper in APME study where the impacts are equivalent.
(44) Results available for PA, PE, PP, ABS, PC, PUR in the APME study.

8.1.6 *Sensitivity analysis about substitution rate*

The environmental impacts and benefits of plastic mechanical recycling greatly depend on the substitution rate as shown by the results/detailed data presented above and in appendix 6 when looking at PUR.

8.1.6.1 *Methodology developed*

In order to assess the environmental impacts and benefits due to mechanical recycling, are considered:

- The impacts generated by the reprocessing with the view to recycling

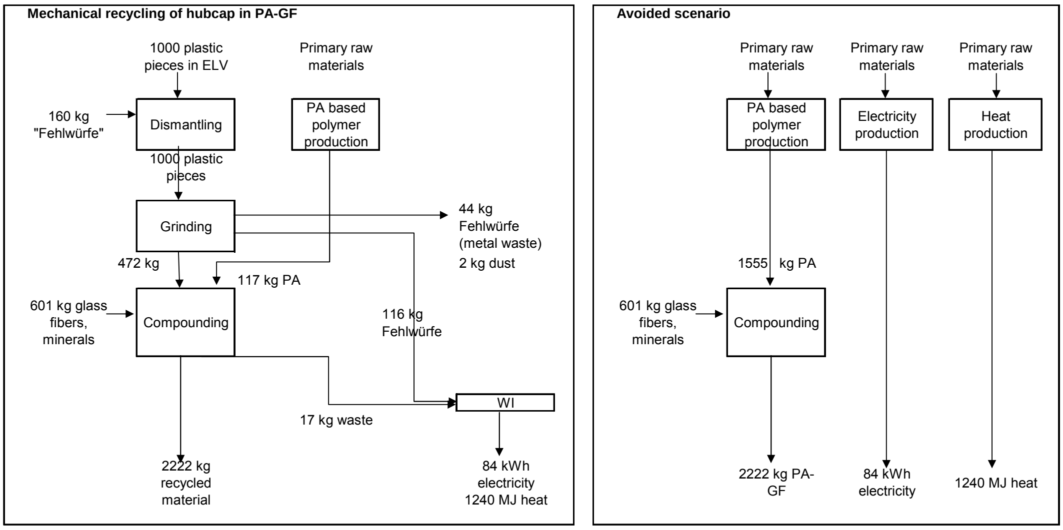
minus

- The impacts linked to the production of primary material

The substitution rate intervenes as follows. Considering for example 1 kg of recyclate and a substitution rate is 0.8, the environmental impacts and benefits are equal to the impacts generated by the recycling process to produce 1 kg of recyclate minus the impacts linked to the production of 0.8 kg of virgin material.

Example

Figure 8.2: Example for the mechanical recycling of PA-GF – SR = 1



Taking the example above, the environmental impacts and benefits of the mechanical recycling of the hubcap with a substitution rate of 1 are:

- The impacts generated by the recycling process of the hubcap

minus

- The impacts linked to the production of electricity
- The impacts linked to the production of heat
- The impacts linked to the production of virgin PA-GF

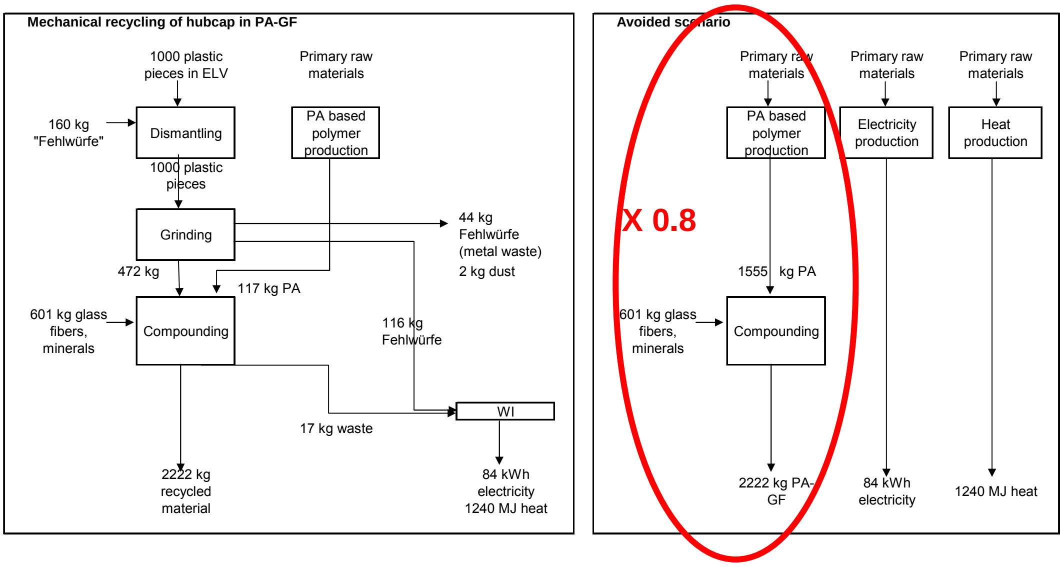
If the substitution rate is only 0.8, the environmental impacts and benefits of the mechanical recycling of the hubcap become:

- The impacts generated by the recycling process of the hubca

minus

- The impacts linked to the production of electricity
- The impacts linked to the production of heat
- 0.8 x The impacts linked to the production of virgin PA-GF

Figure 8.3: Example for the mechanical recycling of PA-GF – SR = 0.8



Calculation of the impacts

Detailed LCA data for each step of the process is only available in the Fraunhofer study. Thus the impacts of mechanical recycling according to different substitution rates were simulated for the following pieces / resins: PA-GF, PUR, PP/EPDM, PVC/PP-TV/ABS/PUR.

For each of these pieces / resins, sensitivity analyses were performed for 5 substitution rates: 0.9, 0.8, 0.7, 0.65 and 0.6.

The environmental impacts were calculated following these steps:

- Step 1: the avoided impacts due to the production of virgin resin were multiplied by the substitution rate
- Step 2: then they were added to other avoided impacts if any (energy spared…) to obtain the new total avoided impacts
- Step 3: the new total avoided impacts was then subtracted to the total generated impacts (recycling process)

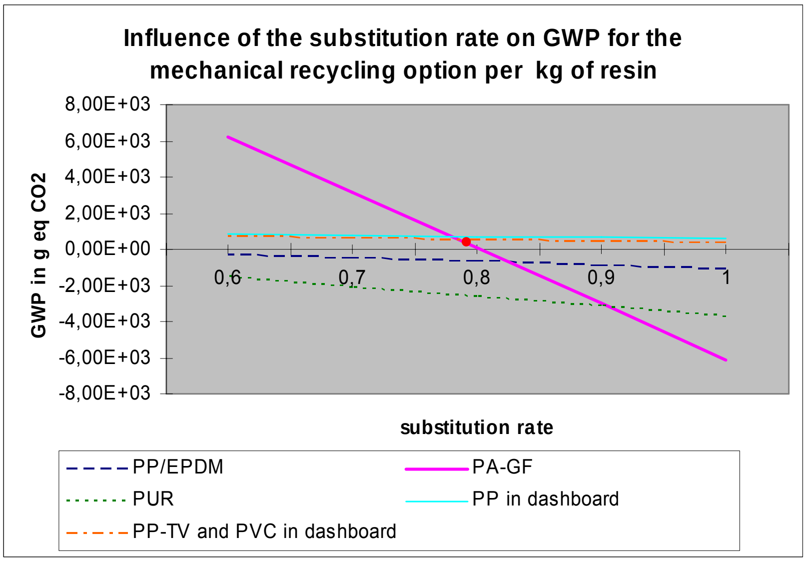
Note that when analysing recycling impacts, the finished products, either from recyclates or from virgin materials, are supposed to fulfil the same function and have similar performances. For a substitution rate of 1, recyclates and virgin granulates have similar performances; thus the boundaries of the system under study usually stop at the entrance of the transformation plant (i.e. does not include the environmental impacts of the transformation stage as they are supposed to be the same in the recycling system and the avoided system). On the contrary, for a substitution rate lower than 1, the transformation stage has theoretically to be taken into account (e.g., energy consumption may be different due to different weights of final products).

For all the resins (except PUR), data were only available for a SR of 1, thus without the transformation stage. For that reason, we were not able to integrate environmental impacts of transformation in the sensitivity analyses performed. The results presented are thus underestimated.

8.1.6.2 Results

Detailed data can be found in appendix 6.

For each impact category, graphs showing the evolution of the impacts of mechanical recycling per resin according to the substitution rate were obtained, such as the graph below for climate change. Similar graphs for the 7 other impact categories can be found in appendix 6.



Two types of information can be deducted by reading the graphs:

- Do substitution rates exist for which the mechanical recycling option is beneficial in terms of environmental impacts per resin and per impact category? It also comes out that when the lower the substitution rate, the lower the environmental benefits or the higher the disbenefits of mechanical recycling. Therefore there sometimes exists a threshold substitution rate below which environmental benefits can even be replaced by disbenefits. This general pattern varies according to the different resins and impact categories.

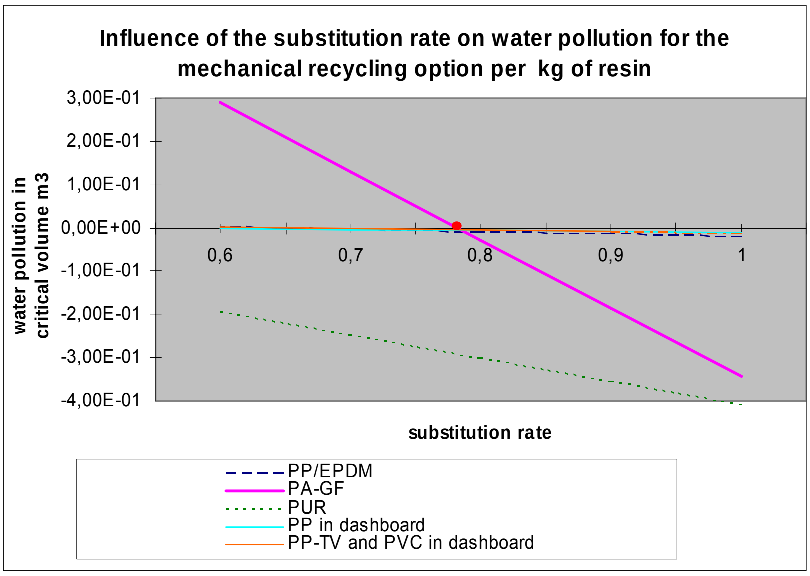
- The result of the comparison of the environmental impacts of the mechanical recycling option versus landfill depends on the substitution rate. There exists a substitution rate for which the environmental impacts and benefits of landfill and mechanical recycling are equivalent. Above this limit, the mechanical recycling option comes out better than landfill; on the contrary, under this limit, landfill comes out better than mechanical recycling. This threshold substitution rate is represented for each resin in the graphs by a red point positioned on the graph of the corresponding resin. The horizontal coordinate of the red point represents the substitution rate for which the environmental impacts and benefits of landfill and mechanical recycling of the concerned resin are equivalent in terms of the impact category considered; the vertical coordinate is the impact value of landfill in terms of the impact category considered.

Note that for some resins and impact categories the environmental impacts of landfill always come out more harmful than mechanical recycling for all impact categories and for all substitution rates. The red points showing the threshold substitution rate for which the environmental impacts and benefits of landfill and mechanical recycling are equivalent are only represented when necessary.

Comments

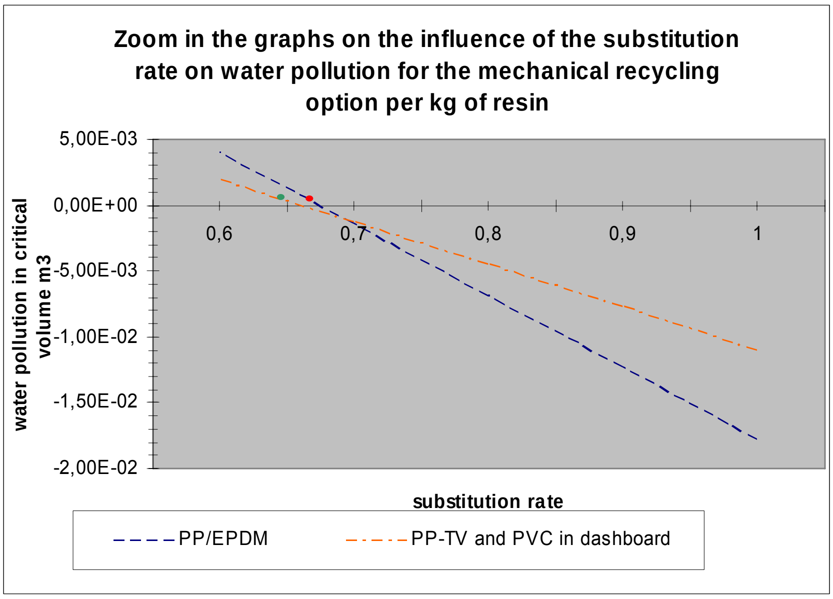
PP/EPDM

- For all impact categories except water pollution, the mechanical recycling of PP/EPDM comes out better than landfill for all substitution rates between 0,6 and 1.
- The horizontal coordinate of the red point on the graph below represents the substitution rate for which the environmental impacts and benefits in terms of water pollution of landfill and mechanical recycling of PP/EPDM are equivalent (the vertical coordinate is the impact value of landfill in terms of water pollution). Above this limit, mechanical recycling of PP/EPDM comes out better than landfill in terms of water pollution; on the contrary, under this limit, landfill comes out better than the mechanical recycling of PP/EPDM in terms of water pollution. Thus for PP/EPDM the mechanical recycling option comes out better than landfill if the substitution rate is above 0,67 approximately.



PP-TV/ABS/PVC/PUR

- For all impact categories except water pollution and hazardous waste, the mechanical recycling of PP from dashboard comes out better than landfill for all substitution rates between 0,6 and 1.
- In terms of hazardous waste, the mechanical recycling of PP from dashboard comes out more harmful than landfill for all substitution rates between 0,6 and 1.
- The horizontal coordinate of the green point on the graph below represents the substitution rate for which the environmental impacts and benefits in terms of water pollution of landfill and mechanical recycling of PP from dashboard are equivalent. Thus for PP from dashboard the mechanical recycling option comes out better than landfill if the substitution rate is above 0,64 approximately; however, the mechanical recycling of PP from dashboard comes out worse than landfill for a substitution rate below 0,64 in terms of water pollution.



- Concerning the impacts of mechanical recycling versus landfill of PP-TV and PVC from dashboard, the results are contrasted. The mechanical recycling of PP-TV and PVC from dashboard comes out better than landfill for all substitution rates above 0,6 in terms of climate change, photochemical oxidation, air acidification and energy consumption; however the mechanical recycling of PP-TV and PVC from dashboard comes out more harmful than landfill for all substitution rates above 0,6 in terms of hazardous waste. For two impact categories (eutrophication, water pollution), there exists a substitution rate above which the mechanical recycling if PP-TV and PVC from dashboard comes out better than landfill and under which landfill comes out better than the mechanical recycling of PP-TV and PVC from dashboard. This limit is approximately 0,61 for eutrophication, and 0,64 for water pollution.

PA-GF

In terms of municipal waste, the mechanical recycling of PA-GF comes out better than landfill for all substitution rates above 0,6. For all other impact categories, a substitution rate exists above which the mechanical recycling of PA-GF comes out better than landfill and under which landfill comes out better than the mechanical recycling of PA-GF. This limit is approximately 0,78 for air acidification, photochemical oxidation, water pollution and energy consumption, 0,75 for eutrophication, 0,8 for climate change, and 0,81 for hazardous waste. (see graphs in appendix 10)

PUR

For all impact categories, the mechanical recycling of PUR comes out better than landfill for all substitution rates between 0,6 and 1. However, the mechanical recycling of a seat cushion in PUR into another seat cushion in PUR with a substitution rate of 0,65 comes out worse than landfill for four impact categories. This shows that when the transformation of the granulates into a finished product is taken into account, the results vary greatly and are more contrasted.

Conclusions

Mechanical recycling can be more beneficial than landfill not only for a substitution rate of 1 (which may be not often reached in real industrial conditions according to plastic experts) **but also for substitution rates lower than 1**. The threshold can not be determined as it varies according to the type of resin and impact category considered (and also because of the limit of the simulations performed: the impacts linked to the transformation of the plastic granulates into a finished product, which were not taken into account by lack of data, may influence greatly the threshold).

8.2 Environmental impacts & benefits associated with different technical options for 2015 targets

8.2.1 Objective of the analysis

In this chapter, the following question is analysed: **In 2015, is it more beneficial to keep the 2006 targets (80% RR / 85% RRR) or to raise them (to 85% RR / 95% RRR or 95% RR / 95% RRR)?**

8.2.2 Methodology developed

Environmental impact quantified

Considering the quantitative Life Cycle Inventory (LCI) data available for each ELV fraction and its different end-of-life options and also the time-budget constraint of the study, we focused on one indicator: Global Warming Potential (GWP). This indicator is most appropriate since it contributes more than 80% to the overall external costs for all treatment options (except for landfill where it contributes to approximately 35-40% of the external costs - see appendix 6).

Scenarios analysed

The GWP was thus calculated for 6 scenarios enabling to reach the 2015 targets or higher targets (95% RR / 95% RRR).

They are similar to those presented in section 5.6 Table 5.11 and based on dismantling or post shredder mechanical separation or post shredder thermal treatment. However, it was decided, for the environmental assessment, to consider a car in 2015 i.e. with a composition as of 2015 to which either 2006 or 2015 targets are applied. As a matter of fact, to answer the question raised in this chapter, it would have been correct, from a methodological point of view, to consider a 2006 composition for the baseline scenario and a 2015 composition for the 2015 scenarios if the change of composition was due to the implementation of 2015 targets.

The following table presents the 6 scenarios assessed.

Table 8.12: Scenarios analysed for the environmental assessment-changes in material treated, 2015 compared to 2006 targets (kg)

Based on Mechanical Separation in 2015												
	vs Dismantling to reach 2006				vs mechanical separation to reach 2006				vs thermal treatment to reach 2006			
Fraction	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill
Ferrous Metal	0	6	0	-6	0	6	0	-6	0	6	0	-6
Non Ferrous Metal	0	2	0	-2	0	1	0	-1	0	0	0	0
Plastics	0	111	-12	-99	0	89	0	-89	0	122	-26	-97
Tyres	0	10	-10	0	0	10	-10	0	0	10	-10	0
Glass	0	-12	0	12	0	0	0	0	0	-5	0	5
Batteries	0	0	0	0	0	0	0	0	0	0	0	0
Fluids	0	0	0	0	0	0	0	0	0	0	0	0
Textiles	0	9	0	-9	0	7	0	-7	0	10	-2	-8
Rubber	0	15	0	-15	0	15	0	-15	0	20	-4	-16
Other	0	0	0	0	0	0	0	0	0	-3	0	3
Total	0	142	-23	-119	0	128	-10	-117	0	161	-42	-119

Based on Thermal Treatment to reach 2015												
	vs Dismantling to reach 2006				vs mechanical separation to reach 2006				vs thermal treatment to reach 2006			
Fraction	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill	Reuse	Recycling	Recovery	Landfill
Ferrous Metal	0	6	0	-6	0	6	0	-6	0	6	0	-6
Non Ferrous Metal	0	2	0	-2	0	1	0	-1	0	0	0	0
Plastics	0	6	62	-68	0	-16	74	-57	0	17	48	-65
Tyres	0	0	0	0	0	0	0	0	0	0	0	0
Glass	0	11	0	-11	0	23	0	-23	0	18	0	-18
Batteries	0	0	0	0	0	0	0	0	0	0	0	0
Fluids	0	0	0	0	0	0	0	0	0	0	0	0
Textiles	0	0	6	-7	0	-2	6	-5	0	1	4	-6
Rubber	0	-2	12	-10	0	-2	12	-10	0	3	8	-11
Other	0	16	0	-16	0	16	0	-16	0	12	0	-12
Total	0	39	80	-119	0	25	92	-117	0	58	60	-119

A 2-step method

Looking at the different scenarios to reach the 2015 targets, differences in flows appear:

- For the following fractions: ferrous metal, non ferrous metal, plastics and process polymer, tyres, glass, textile, rubber, and other
- For the following end-of-life options: reuse, recycling, recovery, landfill

We then developed a 2-step method:

- Step 1: built-up of a database of GWP factors for 1 kg of each concerned fraction and its different end-of-life options
- Step 2: for each scenario, multiplication of these values by the quantities of each flow (fraction / end-of-life option) to quantify the total GWP associated

Figure: A 2-step method

Step 1: Database of GWP factors per kg

Eq. CO_2 / kg	Reuse	Recycling	Recovery	Landfill
Steel	GWP_{S1}	GWP_{S2}		GWP_{S4}
Aluminium	GWP_{A1}	GWP_{A2}		GWP_{A4}
Plastics	GWP_{P1}	GWP_{P2}	GWP_{P3}	GWP_{P4}
Tyres		etc.		
Glass				
Textile				
Rubber				
other				

Legend

	No data necessary because no flow in the scenarios considered

X **Step 2**: Quantification of the GWP for the scenario

Flows for the scenario considered

kg / ELV	Reuse	Recycling	Recovery	Landfill
Steel	S1	S2		S4
Aluminium	A1	A2		A4
Plastics	P1	P2	P3	P4
Tyres		etc.		
Glass				
Textile				
Rubber				
other				

=

GWP obtained for the scenario

Eq. CO_2 / ELV	Reuse	Recycling	Recovery	Landfill	TOTAL
Steel	GWP_{S1} x S1	GWP_{S2} x S2		GWP_{S4} x S4	Σ
Aluminium	etc.				Σ
Plastics					Σ
Tyres					Σ
Glass					Σ
Textile					Σ
Rubber					Σ
other					Σ
TOTAL					**GWP**

Database of GWP factors

Such a database contains the key data to allow the quantification of the environmental impacts associated to a scenario.

We performed a large literature review and selected the best available LCI database or LCA studies to extract the data we were looking for (see appendix 8).

Several decisions and assumptions had to be made (system boundaries, substituted material / products…) in order to build this database.

The database itself is summarised in the table below. Calculation details can be found in annex 6 for plastics recycling and recovery and in annex 7 for the other fractions (as well as plastics reuse). The main hypotheses are listed hereafter.

Table: Database of GWP factors per fraction and end-of-life option

kg eq. CO_2 100 yrs / kg of fraction	Reuse (1)	Recycling	Recovery	Landfill	Source of primary data[66]
Steel	-2,2	-0,39		0 (2)	IISI (1999-2000)
Aluminium	-13	-11		0 (2)	EAA (2000)
Plastics (3)	1,2 to 31 (4)	-6,1 to 4,0	-1,7 to 2,1	0,03 to 0,36	Fraunhofer (2002)
Tyres		-2,7 (5)	-1,2 to 0,68 (6)		PréConsultants B.V. (2001)
Glass (7)		Data n.a.		Data n.a.	
Textile		Data n.a.	Data n.a.	Data n.a.	
Rubber		Data n.a.	Data n.a.	Data n.a.	
Other		Not computable		Not computable	

How to read these data: for example, the reuse of 1 kg of steel avoids the emission of 2,2 kg eq. CO_2 (because it avoids the production of e.g. hot rolled coil from uranium); the recycling of 1 kg of steel avoids the emission of 0,39 kg eq. CO_2 (it corresponds to the impacts generated by the recycling of steel in EAF minus the impacts of the production of steel from uranium which is avoided); landfilling of steel is not known for having a significant impact in terms of greenhouse gases emissions.

(1) Note that the fact that the substituted product is not necessarily 100% from primary material is not taken into account (without being able to say how the results would be affected).
(2) Note that the impacts generated by potential transport to landfill are not taken into account.
(3) Ranges of GWP are given to cover the different types of resins and recovery processes (cement kiln, waste incineration, blast furnace, syngas production). Reliable LCI data are available for PA, PA-GF, PC, PE, ABS, PUR, PP, PP/EPDM, PVC/ABS/PUR/PP which represent only a fraction of the about 100 kg of plastic present in an ELV. **No reliable LCI data** were found in the framework of this study **for the other resins present in ELV including POM, PPE, PVC, PBT, ASA, PMMA, UP**.

66 BIO calculations from primary data extracted from the sources listed in this table

(4) Note that the avoided process is the production of granulates. This is a simplification because granulates transformation to finished products is not taken into account.
(5) This data corresponds to the recycling of tyres as sport surfaces and floors which represents about 45% of the recycling routes in Europe. For information, the other recycling routes are applications in construction as filling materials (22%), additive for bitumen application in road (8%), and consumer goods (24,4%).
(6) This range covers 2 recovery options: waste incineration and cement kiln.
(7) Available data in the literature only refers to packaging glass. According to our contacts with glass recyclers, big differences exist between packaging glass recycling and car glass recycling (incl. the fact that packaging glass is recycled in close-loop whereas car glass is recycled as insulation glass fibres thus substituting insulation material). Despite our efforts, it was not possible to collect quantitative data in the framework of this study.

Based on data available, the system boundaries considered for each end-of-life option are as follows:

Reuse

The environmental impacts and benefits resulting from the reuse of a piece correspond to:

- The impacts generated by the preparation of the piece with the view to reusing it, which are considered negligible

minus

- The impacts linked to the production of an equivalent product from raw material[67]

Recycling

Are considered:

- The impacts generated by the reprocessing with the view to recycling

minus

- The impacts linked to the production of primary material

Recovery

For plastics and tyres, ranges were considered covering different recovery options (cement kiln, waste incineration, and for plastics, also syngas production, blast furnace).

Landfill

No greenhouse gases emissions were considered for metal landfilling.

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67 Note that the fact that the substituted product is not necessarily 100% from the same primary material is not taken into account (without being able to assess how the results would be impacted).

Remark about the plastics fraction:

The GWP of the different end-of-life options of the ELV plastic fraction depends on the type of resin considered. Since the LCI data available relative to the end-of-life of plastics does not cover all the resins present in an ELV (see section 8.1.5), the GWP of each treatment option is given as a range, corresponding to, on one hand, the plastic fraction with the minimum GWP per kg and, on the other hand, the plastic fraction with the maximum GWP per kg. Then, when calculating the GWP for each scenario, the min value (resp. max value) is obtained by assuming that the X kg of plastics to be treated behaves as the resin with the min GWP (resp. max GWP). This is a very simplifying assumption but the only one relevant considering available data.

Representativeness of the results

For each scenario studied to reach the 2015 targets, the representativeness of the results was calculated. In order to reach the 2015 targets, starting from the 2006 targets, the quantity of each ELV fraction which is recovered, recycled or landfilled changes. Unfortunately, data is not available for all of these fractions. The representativeness of the results is the total weight of the fractions for which LCI data is available divided by the total weight of the ELV fractions whose end-of-life scenario changed between 2006 and 2015 (see 5.6 Table 5.11).

8.2.3 Results

Results are presented first for 1 ELV then at the European level.

For each scenario, four types of results are given depending on the recovery option of the plastic fraction: blast furnace (R1), waste incineration (R2), cement kiln (R3), and syngas production (R4). The advantage of distinguishing these four options resides in the fact that ranges obtained for plastics are reduced (and some of them have both min and max either positive or negative). This facilitates the interpretation of the results and the drawing up of conclusions.

The results were calculated using the scenarios presented in section 5.6 Table 5.11.

8.2.3.1 Results to reach the 2015 targets for 1 ELV in 2015

Additional environmental impacts and benefits to reach 2015 targets compared to 2006 targets

environmental indicator considered: Global Warming Potential (GWP) in kg eq. CO2/ELV

2015 targets: RR 95%
RRR 95%

Technology to reach 2015 targets: post-shredder mechanical separation

Scenario 1

	2015 Technology: post-shredder mechanical separation				Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	2006 Technology: dismantling				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	0	6	0	-6	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	0	2	0	-2	-25,4		-25,4		-25,4		-25,4	
Plastics and Process Polymers	0	111	-12	-99	-716,0	445,8	-740,5	438,5	-707,5	463,1	-731,9	444,2
Tyres	0	10	-10	0	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3
Glass	0	-12	0	12	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	0	0	0	0								
Fluids	0	0	0	0								
Textiles	0	9	0	-9	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	0	15	0	-15	data n.a.		data n.a.		data n.a.		data n.a.	
Other	0	0	0	0								
Total	0	142	-23	-119	**-778**	**403**	**-803**	**396**	**-770**	**420**	**-794**	**401**
	Representativeness of the results				78%							
	External costs				**-37 €**	**19 €**	**-39 €**	**19 €**	**-37 €**	**20 €**	**-38 €**	**19 €**

Scenario 2

	2015 Technology: post-shredder mechanical separation				Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	2006 Technology: mechanical separation				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	**0**	**6**	**0**	**-6**	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	**0**	**1**	**0**	**-1**	-6,1		-6,1		-6,1		-6,1	
Plastics and Process Polymers	**0**	**89**	**0**	**-89**	-572,9	352,0	-572,9	352,0	-572,9	352,0	-572,9	352,0
Tyres	**0**	**10**	**-10**	**0**	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3
Glass	**0**	**0**	**0**	**0**								
Batteries	**0**	**0**	**0**	**0**								
Fluids	**0**	**0**	**0**	**0**								
Textiles	**0**	**7**	**0**	**-7**	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	**0**	**15**	**0**	**-15**	data n.a.		data n.a.		data n.a.		data n.a.	
Other	**0**	**0**	**0**	**0**								
Total	**0**	**128**	**-10**	**-117**	**-616**	**328**	**-616**	**328**	**-616**	**328**	**-616**	**328**
	Representativeness of the results				83%							
	External costs				**-30 €**	**16 €**	**-30 €**	**16 €**	**-30 €**	**16 €**	**-30 €**	**16 €**

Scenario 3

	2015 Technology: post-shredder mechanical separation				Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	2006 Technology: thermal treatment				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	0	6	0	-6	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	0	0	0	0								
Plastics and Process Polymers	0	122	-26	-97	-784,1	494,0	-835,6	478,8	-766,3	530,5	-817,5	490,7
Tyres	0	10	-10	0	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3
Glass	0	-5	0	5	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	0	0	0	0								
Fluids	0	0	0	0								
Textiles	0	10	-2	-8	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	0	20	-4	-16	data n.a.		data n.a.		data n.a.		data n.a.	
Other	0	-3	0	3	not computable		not computable		not computable		not computable	
Total	0	161	-42	-119	-821	476	-872	461	-803	513	-854	473
	Representativeness of the results				78%							
	External costs				**-39 €**	**23 €**	**-42 €**	**22 €**	**-39 €**	**25 €**	**-41 €**	**23 €**

Remark regarding scenario 2: results are equal for the 4 sub-scenarios R1, R2, R3 and R4 because plastics are only mechanically recycled in this scenario (no other form of recovery considered).

With respect to the three scenarios considered and the global warming potential (GWP), **to reach the RR 95% / RRR 95% targets** compared to the 2006 targets:

- Available data does not allow determining if GWP will increase or decrease independently from the technology used to reach the targets; indeed, the additional GWP vary in a range (about -870 to +510 kg eq CO_2 per ELV) where the min value is negative (i.e. environmental benefit) and the max value is positive (i.e. environmental impact).
- The impact linked to the plastic fraction is the only determining parameter of the results in terms of GWP (the other materials do not influence significantly the results). Note that no data are available for glass and rubber recycling.
- The higher the quantity of plastics recycled in a scenario, the larger the range of additional environmental impacts and benefits (in scenario 1 with 111 kg of plastic recycled: from about -800 to 420 kg eq CO_2 per ELV; in scenario 2 with 89 kg of plastic recycled: from about -610 to 330 kg eq CO_2 per ELV; in scenario 3 with 122 kg of plastic recycled: from about -870 to 510 kg eq CO_2 per ELV).
- The additional environmental impacts and benefits do not significantly depend on the type of recovery option for plastics. The key parameter is the type of resins. For instance, since the greatest change between 2006 and 2015 in scenario 1 is that 111 kg of plastics (out of 123 kg of plastics present in an ELV in 2015) are recycled, the results greatly depend on the GWP of the recycling of the plastic fractions which varies from -6.1 kg eq CO_2 per kg of PA-GF when the substitution rate is 1 to +4.0 kg eq CO_2 per kg of PUR with a substitution rate of 0.65[68] (see details in appendix 6).

68 Note that this range of results is for the only resins for which LCI data are available.

Additional environmental impacts and benefits to reach 2015 targets compared to 2006 targets

environmental indicator considered: Global Warming Potential (GWP) in kg eq. CO_2/ELV

2015 targets: RR 85%
RRR 95%

Technology to reach 2015 targets: post-shredder Thermal treatment

Scenario 4

	2015 Technology: post-shredder Thermal treatment				Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	2006 Technology: dismantling				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	0	6	0	-6	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	0	2	0	-2	-25,4		-25,4		-25,4		-25,4	
Plastics and Process Polymers	0	6	62	-68	-79,7	29,3	-43,4	151,7	-166,4	-13,2	-71,7	108,6
Tyres	0	0	0	0								
Glass	0	11	0	-11	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	0	0	0	0								
Fluids	0	0	0	0								
Textiles	0	0	6	-7	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	0	-2	12	-10	data n.a.		data n.a.		data n.a.		data n.a.	
Other	0	16	0	-16	not computable		not computable		not computable		not computable	
Total	0	39	80	-119	-107	2	-71	124	-194	-41	-99	81
	Representativeness of the results				63%							
	External costs				**-5 €**	**0 €**	**-3 €**	**6 €**	**-9 €**	**-1 €**	**-5 €**	**4 €**

Scenario 5

	2015 Technology: post-shredder Thermal treatment				Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	2006 Technology: post-shredder mechanical separation				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	**0**	**6**	**0**	**-6**	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	**0**	**1**	**0**	**-1**	-6,1		-6,1		-6,1		-6,1	
Plastics and Process Polymers	**0**	**-16**	**74**	**-57**	-108,0	106,9	-64,4	253,7	-212,0	56,0	-98,4	202,1
Tyres	**0**	**0**	**0**	**0**								
Glass	**0**	**23**	**0**	**-23**	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	**0**	**0**	**0**	**0**								
Fluids	**0**	**0**	**0**	**0**								
Textiles	**0**	**-2**	**6**	**-5**	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	**0**	**-2**	**12**	**-10**	data n.a.		data n.a.		data n.a.		data n.a.	
Other	**0**	**16**	**0**	**-16**	not computable		not computable		not computable		not computable	
Total	**0**	**25**	**92**	**-117**	-116	98	-73	245	-220	48	-107	194
	Representativeness of the results				59%							
	External costs				**-6 €**	**5 €**	**-3 €**	**12 €**	**-11 €**	**2 €**	**-5 €**	**9 €**

Scenario 6

	2015 Technology: post-shredder Thermal treatment				Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	2006 Technology: post-shredder thermal treatment				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	**0**	**6**	**0**	**-6**	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	**0**	**0**	**0**	**0**								
Plastics and Process Polymers	**0**	**17**	**48**	**-65**	-142,4	72,1	-114,1	167,6	-210,1	39,0	-136,2	134,0
Tyres	**0**	**0**	**0**	**0**								
Glass	**0**	**18**	**0**	**-18**	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	**0**	**0**	**0**	**0**								
Fluids	**0**	**0**	**0**	**0**								
Textiles	**0**	**1**	**4**	**-6**	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	**0**	**3**	**8**	**-11**	data n.a.		data n.a.		data n.a.		data n.a.	
Other	**0**	**12**	**0**	**-12**	not computable		not computable		not computable		not computable	
Total	**0**	**58**	**60**	**-119**	-145	70	-116	165	-212	37	-139	132
	Representativeness of the results				60%							
	External costs				**-7 €**	**3 €**	**-6 €**	**8 €**	**-10 €**	**2 €**	**-7 €**	**6 €**

With respect to the three scenarios considered and the global warming potential (GWP), **to reach the RR 85% / RRR 95% targets** compared to the 2006 targets:

- The additional environmental impacts and benefits vary from about -220 to 240 kg eq CO_2 per ELV.
- As in the first set of scenarios (1 to 3), the impact linked to the plastic fraction is the determining parameter of the results in terms of GWP (the other materials do not influence significantly the results). Note that no data are available with respect to glass treatment.
- When considering the recovery options for plastics other than recycling (which are more preponderant in this set of scenario compared to sc 1-2-3 which were more focused on recycling), the same hierarchy can be noticed in the three scenarios: cement kiln better than blast furnace better than syngas production better than waste incineration.
- In one scenario (4), the cement kiln (R3) end-of-life recovery options for plastics is beneficial (emissions of greenhouse gas avoided) in the entire min-max range (-190 to -40 kg eq CO_2 per ELV).

8.2.3.2 *Results to reach the 2015 targets in Europe*

The average number of ELVs in EU-25 in 2004 considered is 10 609 000 ELVs (see table 6 annex 2).

The table below shows the additional environmental impacts and benefits to reach the 2015 targets compared to the 2006 targets for all ELVs in Europe in terms of greenhouse gases emissions.

	Global Warming Potential (GWP) in kt eq. CO2/ELVs in EU-25							
	R1: blast furnace		R2: waste incineration		R3: cement kiln		R4: syngas production	
	Min	Max	Min	Max	Min	Max	Min	Max
Scenario 1	-8257	4273	-8516	4196	-8167	4457	-8425	4256
Scenario 2	-6534	3483	-6534	3483	-6534	3483	-6534	3483
Scenario 3	-8710	5054	-9256	4893	-8521	5441	-9064	5019
Scenario 4	-1140	17	-755	1315	-2059	-434	-1055	858
Scenario 5	-1235	1045	-773	2603	-2339	504	-1133	2054
Scenario 6	-1536	740	-1236	1753	-2253	389	-1470	1397

Remark regarding scenario 2: results are equal for the 4 sub-scenarios R1, R2, R3 and R4 because plastics are only mechanically recycled in this scenario (no other form of recovery considered).

8.2.3.3 *Sensitivity analysis for plastics mechanical recycling*

The objective of this sensitivity analysis is to consider maximum benefits for mechanical recycling.

In the following, the additional environmental impacts and benefits to reach the 2015 targets compared to the 2006 targets for 1 ELV in 2015 were calculated for scenarios 2 and 6 without taking into account the impacts linked to the dashboard composed of 12,5% PVC, 12,5% ABS, 25% PUR, 50% PP-TV nor the results for the recycling of PUR when a substitution rate of 0,65 is considered. Thus the range of GWP factors for the remaining plastics (PP, PUR, PA-GF) for mechanical recycling is -6,1 to -1 kg eq. CO_2 (instead of -6.1 to +4 kg eq. CO_2 when considering all the resins).

The external costs of the environmental benefice were also calculated, considering the following factor: 0.000019-0.000048 Euros/g CO_2 eq (100 yrs) (see §8.1.3).

Additional environmental impacts and benefits to reach 2015 targets compared to 2006 targets
environmental indicator considered: Global Warming Potential (GWP) in kg eq. CO2/ELV

2015 targets: RR 95%
RRR 95%

Technology to reach 2015 targets:
post-shredder mechanical separation

Scenario 2'

	2015 Technology: post-shredder mechanical separation **2006 Technology: mechanical separation**				**Results in kg eq. CO2/ELV** **R1: blast furnace**		**Results in kg eq. CO2/ELV** **R2: waste incineration**		**Results in kg eq. CO2/ELV** **R3: cement kiln**		**Results in kg eq. CO2/ELV** **R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	**0**	**6**	**0**	**-6**	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	**0**	**1**	**0**	**-1**	-6,1		-6,1		-6,1		-6,1	
Plastics and Process Polymers	**0**	**89**	**0**	**-89**	-572,9	-82,6	-572,9	-82,6	-572,9	-82,6	-572,9	-82,6
Tyres	**0**	**10**	**-10**	**0**	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3	-34,5	-15,3
Glass	**0**	**0**	**0**	**0**								
Batteries	**0**	**0**	**0**	**0**								
Fluids	**0**	**0**	**0**	**0**								
Textiles	**0**	**7**	**0**	**-7**	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	**0**	**15**	**0**	**-15**	data n.a.		data n.a.		data n.a.		data n.a.	
Other	**0**	**0**	**0**	**0**								
Total	**0**	**128**	**-10**	**-117**	**-616**	**-106**	**-616**	**-106**	**-616**	**-106**	**-616**	**-106**
	Representativeness of the results				83%							
	External costs				**-30 €**	**-2 €**	**-30 €**	**-2 €**	**-30 €**	**-2 €**	**-30 €**	**-2 €**

With respect to the scenario considered and the global warming potential (GWP), **to reach the <u>RR 95%</u> / RRR 95% targets** compared to the 2006 targets, results are greatly **beneficial** (negative value, avoided impacts) considering that the 87 kg of plastic resins recycled in this scenario behave like PP, PA-GF, or PUR. The **additional environmental benefits** vary **from about -620 to -110 kg eq CO_2 per ELV**. Considering the external costs, such figures enable to **save 2 to 30 Euros/ELV**.

Additional environmental impacts and benefits to reach 2015 targets compared to 2006 targets

environmental indicator considered: Global Warming Potential (GWP) in kg eq. CO2/ELV

2015 targets: RR 85%
RRR 95%

Technology to reach 2015 targets: post-shredder Thermal treatment

Scenario 6'

	2015 Technology: post-shredder Thermal treatment				**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**	
	2006 Technology: post-shredder thermal treatment				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	**0**	**6**	**0**	**-6**	-2,3		-2,3		-2,3		-2,3	
Non Ferrous Metal	**0**	**0**	**0**	**0**								
Plastics and Process Polymers	**0**	**17**	**48**	**-65**	-142,4	-12,2	-114,1	83,2	-210,1	-45,3	-136,2	49,6
Tyres	**0**	**0**	**0**	**0**								
Glass	**0**	**18**	**0**	**-18**	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	**0**	**0**	**0**	**0**								
Fluids	**0**	**0**	**0**	**0**								
Textiles	**0**	**1**	**4**	**-6**	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	**0**	**3**	**8**	**-11**	data n.a.		data n.a.		data n.a.		data n.a.	
Other	**0**	**12**	**0**	**-12**	not computable		not computable		not computable		not computable	
Total	**0**	**58**	**60**	**-119**	-145	-15	-116	81	-212	-48	-139	47
	Representativeness of the results				60%							
	External costs				**- 7 €**	**- 1 €**	**- 6 €**	**4 €**	**- 10 €**	**- 2 €**	**- 7 €**	**2 €**

- The additional environmental impacts and benefits vary from about -210 to 80 kg eq CO_2 per ELV.
- Two end-of-life recovery options for plastics (R1: blast furnace, R3: cement kiln) are beneficial (emissions of greenhouse gas avoided) in the entire min-max range (-210 to -10 kg eq CO_2 per ELV).
- In scenario 6', the external costs range from the saving of 10 Euros/ELV to the cost of 4 Euros/ELV. If only the two end-of-life recovery options which are beneficial are considered (R1: blast furnace, R3: cement kiln), 1 to 10 Euros/ELV are saved.

8.2.3.4 *Focus on recycling targets*

The objective is here to present environmental impacts and benefits of different recycling (and re-use) targets for a 2015-composition car.

Based on possible technologies, the following recycling (and re-use) targets were considered: 78% (current recycling rate), 81%, 85%, 90% and 95%.

The following tables present the scenarios and the GWP obtained for recycling. In order to analyse the benefits of diverting from landfill, results are also given for landfill.

Caveats: once again, these results are only based on data for about 15-20% of plastics out of the total plastics contained in an ELV in 2015. According to plastics experts, the environmental profile of the recycling of these other plastics would be not as good (or worse) than those assessed. For that reason, **the benefits presented below are overestimated and the impacts underestimated**.

Table 8.12: Environmental benefits for different recycling (and re-use) targets

	Baseline/market + depollution - with 2015 ELV (kg)			Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	RR 78%			**Reuse and Recycling**		**Rest to landfill**	
	Reuse	Recycling	Rest	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	31	620	15	-310.5		0.0	
Non Ferrous Meta	9	78	5	-982.5		0.0	
Plastics	1	0	122	-19.1	-0.7	3.8	44.1
Tyres	10	10	10	data n.a.		data n.a.	
Glass	0	0	30	data n.a.		data n.a.	
Batteries	1	12	0	data n.a.		data n.a.	
Fluids	5	12	0	data n.a.		data n.a.	
Textiles	0	0	10	data n.a.		data n.a.	
Rubber	0	0	21	data n.a.		data n.a.	
Other	0	0	21	not computable		not computable	
Total	58	733	234	-1312	-1294	4	44

	2006 target (PST route - thermal treatment)			Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	RR 81,3%			**Reuse and Recycling**		**Rest to landfill**	
	Reuse	Recycling	rest	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	35	655	7	-332.2		0.0	
Non Ferrous Meta	8	74	0	-918.4		0.0	
Plastics	1	0	103	-31.8	-1.2	3.2	36.9
Tyres	10	10	10	data n.a.		data n.a.	
Glass	1	5	26	data n.a.		data n.a.	
Batteries	1	12	0	data n.a.		data n.a.	
Fluids	5	12	0	data n.a.		data n.a.	
Textiles	0	0	10	data n.a.		data n.a.	
Rubber	0	0	21	data n.a.		data n.a.	
Other	0	3	17	not computable		not computable	
Total	61	772	193	-1282	-1252	3	37

	2015 target (PST route - thermal treatment)			Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	RR 85%			**Recycling**		**Rest to landfill**	
	Reuse	Recycling	Rest	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	35	662	0	-334.9		0.0	
Non Ferrous Meta	8	74	0	-918.4		0.0	
Plastics	1	14	88	-119.3	56.2	2.7	31.7
Tyres	10	10	10	data n.a.		data n.a.	
Glass	1	23	7	data n.a.		data n.a.	
Batteries	1	12	0	data n.a.		data n.a.	
Fluids	5	12	0	data n.a.		data n.a.	
Textiles	0	1	9	data n.a.		data n.a.	
Rubber	0	3	18	data n.a.		data n.a.	
Other	0	16	5	not computable		not computable	
Total	61	828	137	-1373	-1197	3	32

	2015 target (PST route - reduced thermal treatment)			Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	RR 90%			**Recycling**		**Rest to landfill**	
	Reuse	Recycling	Rest	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	35	662	7	-334.9		0.0	
Non Ferrous Meta	8	74	0	-918.4		0.0	
Plastics	1	0	103	-31.8	-1.2	3.2	36.9
Tyres	10	10	10	data n.a.		data n.a.	
Glass	1	5	26	data n.a.		data n.a.	
Batteries	1	12	0	data n.a.		data n.a.	
Fluids	5	12	0	data n.a.		data n.a.	
Textiles	0	0	10	data n.a.		data n.a.	
Rubber	0	0	21	data n.a.		data n.a.	
Other	0	3	17	not computable		not computable	
Total	61	779	193	-1285	-1255	3	37

	2015 target (PST route - mechanical separation)			Results in kg eq. CO2/ELV		Results in kg eq. CO2/ELV	
	RR 95%			**Recycling**		**Rest to landfill**	
	Reuse	Recycling	Rest	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	35	662	0	-334.9		0.0	
Non Ferrous Meta	8	74	0	-918.4		0.0	
Plastics	1	103	0	-657.0	408.8	0.0	0.0
Tyres	10	21	0	data n.a.		data n.a.	
Glass	1	0	30	data n.a.		data n.a.	
Batteries	1	12	0	data n.a.		data n.a.	
Fluids	5	12	0	data n.a.		data n.a.	
Textiles	0	10	0	data n.a.		data n.a.	
Rubber	0	20	0	data n.a.		data n.a.	
Other	0	0	20	not computable		not computable	
Total	61	914	51	-1910	-845	0	0

The following graphs summarise these results: first for recycling alone then for the difference between recycling versus landfill.

Figure 8.4: Environmental benefits for different recycling (and re-use) targets

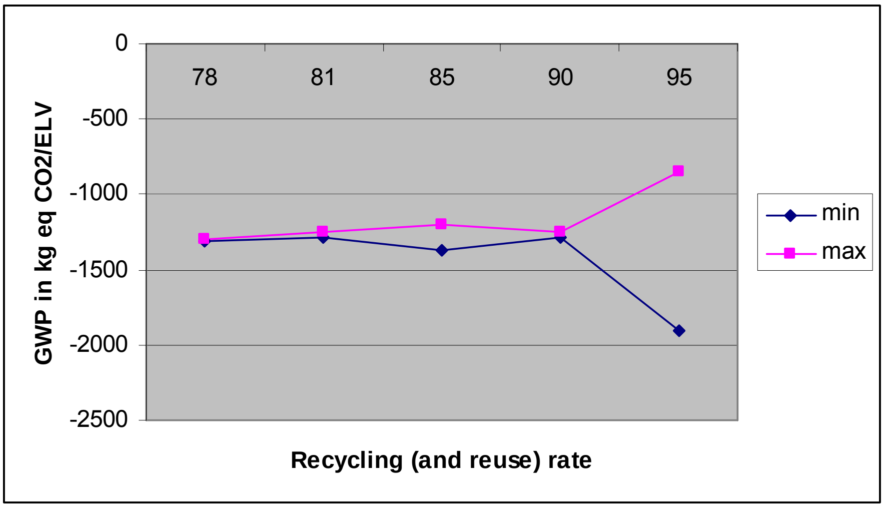
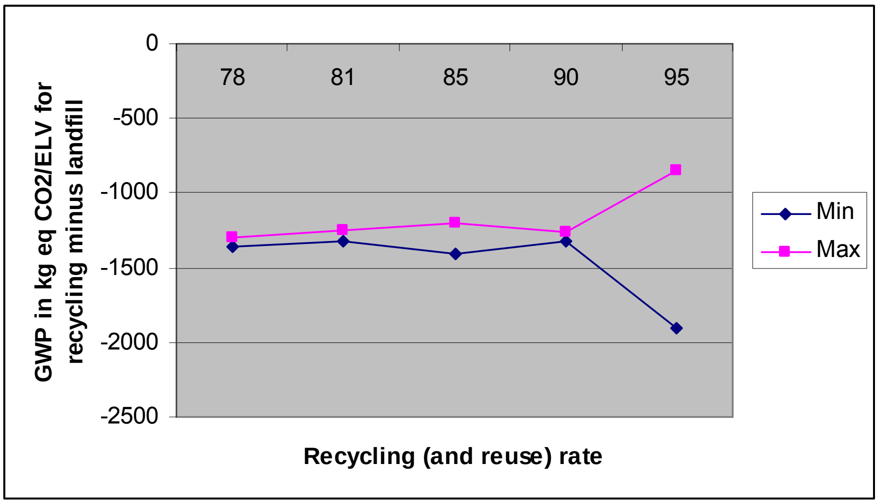


Figure 8.5: Net environmental benefits for different recycling (and re-use) targets when comparing recycling to landfill



Remark: the 2 graphs do not differ significantly as the impacts generated by landfill which are subtracted to recycling are with an order of magnitude much smaller compared to the direct benefits of recycling.

As a conclusion:

- ELV recycling is environmentally beneficial for GWP whatever the recycling (and re-use) target between 78% and 95%. This is mainly due to the positive contribution of ferrous and non ferrous metals recycling.
- These benefits are even (a little bit) higher when the avoided impacts of landfill are added.
- The higher the recycling rate, the lower the environmental benefits. This is due to the fact that plastics have to be recycled and among those plastics, not all of them have a beneficial GWP for recycling.

Thus **if high recycling targets were to be set up, then the environmental benefits of metal recycling would largely compensate the potential environmental disbenefits of some plastics recycling**.

8.2.4 General conclusions

Caveats: these conclusions concern mainly global warming potential GWP (which were found in this study to explain more than 80% of total external costs). They may be different for the other environmental impacts. Besides, no data were available for all materials, in particular glass and rubber.

Recycling targets

- The current situation (about 78% recycling rate) is environmentally beneficial for GWP. This is mainly due to the positive contribution of ferrous and non ferrous metals recycling (strong bonus from avoiding producing hot rolled steel coil from uranium and aluminium ingot from bauxite).

  When considering the diversion from landfill, these benefits are even higher (because of the avoided impacts from landfill).

- Compared to the current situation (about 78% recycling rate), 80% recycling rate (current 2006 target) can involve an increase of environmental benefits.

  It actually depends on the additional fractions which are recycled:

  - additional metals recycled would be beneficial;
  - the recycling of some easy recyclable plastics would also be beneficial (for some of them, a high substitution rate close to 1 would however be a necessary condition – a specific study would be useful to further identify the feasibility);
  - if another option is chosen on the ground (for instance glass recycling), the outcome can not be predicted as no LCA data are available.

- If higher recycling (and re-use) targets were to be set up, then the environmental benefits of metal recycling would largely compensate the potential environmental disbenefits of other fractions recycling, including of some plastics. And recycling (and reuse) would still result in a net environmental benefit (due to both the diversion from landfill and the avoidance of impacts linked to the production from virgin materials).

- Above a certain threshold (which is not possible to determine but which is higher than 78%, i.e. the current situation with no plastic recycled), the higher the recycling target, the lower the environmental benefits. This is due to the fact that plastics have to be recycled and among those plastics, not all of them have a beneficial GWP for recycling.

  Compared to 2006 target, an increase of the recycling target would mean a higher proportion of plastics to be recycled. The environmental profile of plastics recycling is highly dependant on different key parameters: type of resin and type of resin mix, level of substitution rate. In the most favourable conditions (high substitution rate and resin not too difficult to recycle69), recycling is environmentally beneficial. In the other cases, recycling generate impacts that avoided impacts from landfill do not compensate.

- As a result, from an environmental point of view, there is no clear environmental justification for recycling targets higher than 80% in the ELV directive except if one accepts that environmental benefits brought by some materials can compensate the potential disbenefits linked to other materials.

Recovery targets

- Depending on the recovery option and recovery process characteristics, the expected GWP of higher recovery targets can increase or decrease compared to 2006.
- The impact linked to the plastic fraction is the only determining parameter of the results in terms of GWP (the other materials do not influence significantly the results).

The higher the quantity of plastics recycled in a scenario, the higher the uncertainty about the nature (benefit or disbenefit) and the level of the impact. There are two main reasons for that:

– The fact that the environmental profile of plastics recycling may vary a lot (according to resins and substitution rates for instance).

– When considering the recovery options for plastics other than recycling, there are cases when recovery is more beneficial compared to landfill but not all (depending mainly on the type and quantity of substituted resources).

It would probably be possible to identify some specific treatment options and characteristics for which plastics recovery would be beneficial compared to landfill (e.g. MSWI with high efficiency rate and substitution of an energy mix relatively polluting, cement kiln with brown coal substitution, syngas with waste oil substitution). But further analysis (in particular to cover other types of plastics resins, local conditions and technology characteristics) would be necessary.

8.3 Environmental impacts & benefits associated with 2006 targets

8.3.1 Objective of the analysis

In this chapter, the following question is analysed: **What are the environmental impacts and benefits of the 2006 targets (80% RR / 85% RRR) versus the market baseline recycling rates for an ELV in 2006 (80,1% RR / 81,1% RRR)?**

8.3.2 Methodology developed

The methodology is the same as the one described above for the 2015 targets (see §8.2.2).

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69 As a reminder, the resins studied in the available LCAs are those considered by plastics experts as the easiest recyclable. The others are expected not to have a better environmental profile.

The additional environmental impacts and benefits to reach the 2006 targets compared to the market baseline targets are considered looking at a car in 2006 (i.e. with a car composition as of 2006).

The Global Warming Potential (GWP) was calculated for each scenario enabling to reach the 2006 targets (dismantling, post shredder mechanical separation, post shredder thermal treatment).

8.3.3 Results

Here also results are presented first for 1 ELV then at the European level.

8.3.3.1 Results to reach the 2006 targets for 1 ELV in 2006

Additional environmental impacts and benefits to reach 2006 targets compared to market baseline

environmental indicator considered: Global Warming Potential (GWP) in kg eq. CO_2/ELV

2006 targets: RR 80% / RRR 85%

Scenario 7

	2006 Technology: dismantling				**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**	
	RR: 83% RRR: 85%				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	-2	9	0	-7	0,9		0,9		0,9		0,9	
Non Ferrous Metal	0	1	0	-2	-12,4		-12,4		-12,4		-12,4	
Plastics and Process Polymers	0	9	10	-18	-64,5	36,5	-58,6	56,4	-78,6	29,6	-63,2	49,4
Tyres	0	0	0	0								
Glass	0	12	0	-12	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	0	0	0	0								
Fluids	0	1	0	0	data n.a.		data n.a.		data n.a.		data n.a.	
Textiles	0	1	0	-1	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	0	5	0	-5	data n.a.		data n.a.		data n.a.		data n.a.	
Other	0	0	0	0								
Total	-2	38	10	-45	-76	25	-70	45	-90	18	-75	38
	Representativeness of the results				61%							
	External costs				**-4 €**	**1 €**	**-3 €**	**2 €**	**-4 €**	**1 €**	**-4 €**	**2 €**

Scenario 8

	2006 Technology: post-shredder mechanical separation				**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**	
	RR: 84% RRR: 85%				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	**-2**	**9**	**0**	**-7**	0,9		0,9		0,9		0,9	
Non Ferrous Metal	**0**	**3**	**0**	**-2**	-33,0		-33,0		-33,0		-33,0	
Plastics and Process Polymers	**0**	**26**	**0**	**-26**	-168,0	103,2	-168,0	103,2	-168,0	103,2	-168,0	103,2
Tyres	**0**	**0**	**0**	**0**								
Glass	**0**	**0**	**0**	**-1**	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	**0**	**0**	**0**	**0**								
Fluids	**0**	**1**	**0**	**0**	data n.a.		data n.a.		data n.a.		data n.a.	
Textiles	**0**	**3**	**0**	**-3**	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	**0**	**5**	**0**	**-5**	data n.a.		data n.a.		data n.a.		data n.a.	
Other	**0**	**0**	**0**	**0**								
Total	**-2**	**47**	**0**	**-44**	-200	71	-200	71	-200	71	-200	71
	Representativeness of the results				83%							
	External costs				**-10 €**	**3 €**	**-10 €**	**3 €**	**-10 €**	**3 €**	**-10 €**	**3 €**

Scenario 9

	2006 Technology: post-shredder thermal treatment				**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**		**Results in kg eq. CO2/ELV**	
	RR: 81,4% RRR: 85%				**R1: blast furnace**		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
Fraction	**Reuse**	**Recycling**	**Recovery**	**Landfill**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Ferrous Metal	-2	9	0	-7	0,9		0,9		0,9		0,9	
Non Ferrous Metal	0	3	0	-2	-33,0		-33,0		-33,0		-33,0	
Plastics and Process Polymers	0	0	20	-20	-13,0	1,6	-1,2	41,4	-41,2	-12,2	-10,4	27,4
Tyres	0	0	0	0								
Glass	0	5	0	-5	data n.a.		data n.a.		data n.a.		data n.a.	
Batteries	0	0	0	0								
Fluids	0	1	0	0	data n.a.		data n.a.		data n.a.		data n.a.	
Textiles	0	0	2	-2	data n.a.		data n.a.		data n.a.		data n.a.	
Rubber	0	0	4	-4	data n.a.		data n.a.		data n.a.		data n.a.	
Other	0	3	0	-3	not computable		not computable		not computable		not computable	
Total	-2	21	26	-43	-45	-31	-33	9	-73	-44	-43	-5
	Representativeness of the results				70%							
	External costs				**-2 €**	**-1 €**	**-2 €**	**0 €**	**-4 €**	**-1 €**	**-2 €**	**0 €**

Remark regarding scenario 8: results are equal for the 4 sub-scenarios R1, R2, R3 and R4 because plastics are only mechanically recycled in this scenario (no other form of recovery considered).

With respect to the three scenarios considered and the global warming potential (GWP), **to reach the RR 80% / RRR 85% targets** compared to the market baseline:

- Available data does not allow to determine if GWP will increase or decrease independently from the technology used to reach the targets; indeed, the additional GWP vary in a range (about -200 to +70 kg eq CO_2 per ELV) where the min value is negative (i.e. environmental benefit) and the max value is positive (i.e. environmental impact).
- The impact linked to the **plastic fraction is the only determining parameter** of the results in terms of GWP (the other materials do not influence significantly the results). Note that no data are available for glass and rubber recycling.
- With respect to the three technologies considered to reach the 2006 targets, **the post-shredder thermal treatment is beneficial (emissions of greenhouse gas avoided) when considering 3 of the 4 plastics recovery options (cement kiln, blast furnace and syngas production)**, in the entire min-max range (-75 to -45 kg eq CO_2 per ELV for cement kiln; -45 to -30 kg eq CO_2 per ELV for blast furnace; -45 to -5 kg eq CO_2 per ELV for syngas production).
- For the two other technologies considered to reach the 2006 targets (**dismantling and post-shredder mechanical separation**), it is **not possible to conclude** because the additional GWP vary in a range (about -200 to +70 kg eq CO_2 per ELV) where the min value is negative (i.e. environmental benefit) and the max value is positive (i.e. environmental impact). Also, note that **the higher the quantity of plastics recycled in a scenario, the larger the range of additional environmental impacts and benefits** (in scenario 7, with 9 kg of plastic recycled: from about -90 to +45 kg eq CO_2 per ELV; in scenario 8, with 26 kg of plastic recycled: from about -200 to +70 kg eq CO_2 per ELV).

  For these two technologies (dismantling and post-shredder mechanical separation), the additional environmental impacts and benefits are **much more influenced by the type of resins than by the plastics recovery option considered**[70].
- When considering the recovery options for plastics other than recycling, a clear hierarchy arises in the studied scenarios: **cement kiln better than blast furnace better than syngas production better than waste incineration**.

70 For instance, with respect to scenario 7 (dismantling), the GWP of the plastic recycling varies from -6,1 kg eq CO_2 per kg of resin for PA-GF when the substitution rate is 1 to +4,0 kg eq CO_2 per kg of resin for PUR with a substitution rate of 0,65. Therefore the GWP of the recycling of 9 kg of plastics in 2006 vary between -54,9 to +36 kg eq CO_2 per ELV. The results of the additional environmental impacts and benefits to reach the 2006 targets (RR 80% / RRR 85%) compared with the market baseline targets are thus mostly due to the impacts of the recycling of 9 kg of plastics: they are beneficial in terms of greenhouse gases emissions for all plastic recovery options for PP/EPDM, PA-GF, PC, ABS, PE, PP, PA, PP-TV from the dashboard or PUR recycled with a substitution rate equal to 1; however the results are harmful in terms of greenhouse gases emissions for PUR recycled with a substitution rate of 0,65. The results are harmful in terms of greenhouse gases emissions for both the PP-TV and the PVC and the particle plate from the dashboard recycled for all end-of-life plastic recovery options but one (R3: cement kiln).

8.3.3.2 Results to reach the 2006 targets in Europe

Based on an average number of ELVs in EU-25 in 2004 of 10,609,000 ELVs, the table below shows the additional environmental impacts and benefits to reach the 2006 targets compared to the market baseline targets for all ELVs in Europe in terms of greenhouse gases emissions.

	Global Warming Potential (GWP) in kt eq. CO2/ELVs in EU-25							
	R1: blast furnace		**R2: waste incineration**		**R3: cement kiln**		**R4: syngas production**	
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Scenario 7	-806	266	-743	477	-955	193	-792	403
Scenario 8	-2123	754	-2123	754	-2123	754	-2123	754
Scenario 9	-479	-324	-353	98	-778	-470	-451	-50

Remark regarding scenario 8: results are equal for the 4 sub-scenarios R1, R2, R3 and R4 because plastics are only mechanically recycled in this scenario (no other form of recovery considered).