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Comparative Study of End-of-Life Vehicle Recycling in Australia and Belgium

Vi Kie Soo[a]*, Jef Peeters[b], Paul Compston[a], Matthew Doolan[a], Joost R. Duflou[b]

[a] *Research School of Engineering, College of Engineering and Computer Science, The Australian National University, Canberra, ACT 2601, Australia*

[b]*Centre for Industrial Management, KU Leuven, Celestijnenlaan 300A Box 2422, Leuven 3001, Belgium*

* Corresponding author. Tel.: +6-126-125-5941; fax: +6-126-125-0506. *E-mail address:* vikie.soo@anu.edu.au

Abstract

The increasing complexity of multi-material vehicle designs has created challenges for vehicle recycling. Many countries have implemented different end-of-life vehicles (ELVs) treatment policies and guidelines. For example, the European Commission has set recycling and recovery targets for end-of-life vehicle (ELV). This paper discusses a comparative study on the legislative boundaries and environmental performance of the current ELV recycling processes analysed between recycling companies in Australia and Belgium. It is shown that the strict implementation of the ELV Directive in Belgium has led to better environmental performance, by a factor of 7.9 in comparison to the Australian scenario. The enactment of strict ELV legislation, adoption of advanced recycling technologies, and improvement of the recycling efficiencies of revenue streams are identified as the major influencing factors for a sustainable ELV management system.

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Keywords: end-of-life vehicles; vehicle recycling; Life Cycle Assessment; recycling efficiency

1. Introduction

The waste produced by the growing number of vehicles reaching end-of-life (EoL) has been a global concern due to its environmental impact. In 2010, there were about 40 million end-of-life vehicles (ELVs) globally [1]. ELVs are managed differently for different countries. Australia has no formal legislation specifically for end-of-life vehicle (ELV) disposal whereas Belgium enforces the strict ELV Directive of 95% reuse and recovery. In 2010, the numbers of deregistered vehicles in Australia and Belgium were about 600,000 units [1] and 400,000 units [2] respectively. From these numbers, only about 500,000 units [1] and 200,000 units [3] of ELVs were treated within the aforementioned countries.

The adoption of different ELV management systems can lead to different EoL treatment strategies. In Belgium, the strict legislative framework outlined in the ELV Directive has forced recyclers to progressively improve their processes and ensures vehicle manufacturers take responsibility for the EoL treatment of their products. In this context, automotive shredder residue (ASR) has been targeted for further recycling of valuable metals and non-metallic materials to meet the strict legislation. On the contrary, there are only voluntary based ELV recycling guidelines for Australian recyclers that are based on the European Union's ELV Directive. This leads to ASR entering landfill without further treatment to reduce recycling cost.

The choice of EoL treatment strategies has a major influence on the ELV environmental performance and recycling costs. For many years, high steel content in ELV has made them attractive to be acquired by recyclers. Shredding and magnetic separation are commonly used to retrieve steel with high efficiency and low cost. However, the increasing use of lightweight materials in vehicle design has led to the importance of recovering other materials such as plastics. In Belgium, the market for high quality secondary plastics is developed, and has encouraged recyclers to improve their post-shredder treatment technologies while restricted by the recycling costs. The lack of market for secondary plastics in Australia has discouraged further ASR treatment.

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In this paper, a comparative study of ELV recycling in Australia and Belgium is carried out from the perspective of legislative framework and environmental performance. The recycling system in Belgium is reflective of the European scenario. Industry data collected from these two countries allow comparison for ELV management systems adopted in different regions. The findings are essential for understanding the barriers and opportunities to improve material recycling for both countries. The information can be used by policy makers to deploy effective actions to improve the current recycling practices for the respective countries.

2. ELV Regulatory Framework

The management of ELV waste is restricted by a wide variety of national legislations. Countries and regions such as the European Union, Japan and Korea have specific ELV related legislation to manage waste disposal. However, certain industrialised countries with high vehicle penetration rate, such as Australia, Canada and the US, have no specific mandatory legislation [1].

2.1. Australia

The ELV management in Australia is driven by economic mechanisms, with no existing national legislation related to ELV disposal [4,5]. ELVs are acquired by recyclers due to the value of metal scrap, and they are responsible for the disposal of ELV waste at their own expense. The amount of waste generated from ELVs is significant and can be costly.

Despite the lack of ELV legislation in Australia, the disposal of certain toxic substances is captured under different and more broadly defined voluntary product stewardship arrangements bound by the Product Stewardship Act 2011 [6]. Voluntary product stewardship involves parties voluntarily seeking accreditation for their product stewardship arrangement from the Australian Government, as is the case for the Australian Battery Recycling Initiative, the Product Stewardship for Oil Program, and the Tyre Stewardship Australia [7–9]. Therefore, the recycling of certain vehicle parts, such as batteries, fluids, and tyres, are captured under these organisations. The National Waste Policy is responsible for the product stewardship framework [10].

One of the major consequences arising from voluntary based waste policy is the competition between legitimate and illegitimate recycling sectors. The illegitimate recycling sectors do not adhere to the environmental standards, and often provide competitive prices during the ELV collection process due to their low recycling costs [4]. This has consequently led to the disposal of large amounts of ELV waste without proper treatment.

About 25% of the ELV is ASR that ends up in landfills [11]. ASR landfills contain hazardous waste that is constrained by the landfill standards covered in the waste management strategies [12]. A landfill levy is imposed to deter landfill and promote alternative waste treatment options that increase material recycling such as plastics [13,14]. Nevertheless, the landfill costs are still low in comparison to other countries [15].

2.2. Belgium (European Union)

The ELV management system in Belgium is driven by ELV Directive 200/53/EC enacted in the year 2000 [16]. It covers different aspects involving all parties from vehicle production to recycling stages based on the subsidiarity principle [17] and extended producer responsibility policy [1]. The subsidiarity principle is defined as the fulfillment of the Directive’s guidelines based on individual approaches of the Member States in their countries [17]. This has led to slight differences in the approach taken to comply with the regulatory requirements [18].

In Belgium, the ELV Directive is implemented at regional level, and monitored by Febelauto, a non-profit organisation. Febelauto manages the collection, treatment and recycling of ELV. They also inform and support different parties involved in the ELV management system, such as last vehicle owners, recycling operators, authorised treatment facilities, and authorities [19].

The most pertinent legislation to vehicle recyclers are the strict quantified targets to be achieved for reuse, recycling and recovery of ELV. Recycling refers to the retrieval of waste materials for reuse in a closed-loop or open-loop system, whereas recovery refers to the use of waste materials to generate energy. As shown in Equation 1 and 2, recycling and recovery efficiencies (η) are defined as the total mass (m) of material output from the recycling processes, either for reuse or energy recovery, divided by the input, taking into consideration material losses during processing. Based on the ELV Directive, by 2015 85% of ELV mass needs to be reused and recycled. A further 10% can be used in energy recovery [20]. Therefore, the targets for reuse and recovery combined amount to 95% by mass [16]. This has consequently pressured vehicle recyclers to continuously improve their recycling techniques and post-shredder treatment technologies while generating revenue for their companies. Moreover, the amount of ASR landfilled has decreased and minimised due to the lack of landfill space, charges for landfill disposal, and strict landfill waste legislation [21,22].

$$\eta_{recycling} = \frac{m_{output\_recycling\_process}}{m_{input\_process}} \tag{1}$$

$$\eta_{recovery} = \frac{m_{output\_recovery}}{m_{input\_process}} \tag{2}$$

3. ELV Flow

A generic ELV flow from the vehicle’s last owner to the recycling phase is shown in Figure 1. The collected ELVs undergo depollution procedures to remove batteries, fluids and other materials that contain hazardous waste. Valuable parts are further disassembled to cater for the sale of reuse parts. The depolluted car hulks are then processed in material recycling facilities to recover valuable materials such as ferrous (Fe) and non-ferrous (NF) metals. In countries with strict compliance to

ELV legislations, the remaining ASR is further treated through post-shredder technologies to achieve the set recycling targets.

3.1. Differences in ELV Collection and Recycling Systems

One of the major differences during the collection stage in Australia and Belgium is the issue of certificate of destruction for ELV. This requirement is carried out to ensure ELVs are collected and disposed lawfully through an authorised recycling facility [23]. The number of ELVs collected into proper recycling facilities has an impact on the cost effectiveness of material recycling processes and further post-shredder treatments. As seen in the Australian scenario, the lack of a proper collection system gives opportunities for unauthorised recycling facilities to compete with legitimate recycling sectors in acquiring ELVs [4].

There is a lack of initiative among Australian legitimate recycling facilities to invest in better recycling technologies since they do not receive large volumes of ELV. In Australia, basic recycling processes are used in comparison to the more rigorous recycling technologies adopted in Belgium (Europe). The continuous development of high performance recycling processes, such as density media separation and energy recovery facilities, enables further retrieval of valuable materials and thus, reduces the amount of waste to be landfilled in Europe.

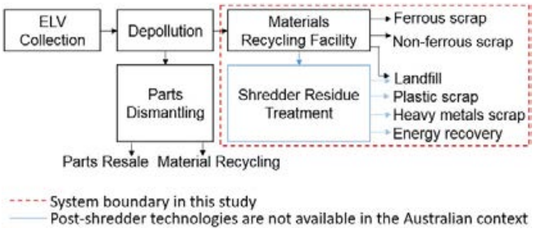

Figure 1: Generic ELV process flow.

3.2. ELV Regulatory Impact on Treatment Strategies

In Australia, the voluntary based ELV regulatory framework has led to a profit-driven automotive recycling industry. The types of recovered materials are limited to high volume metals with low recovery cost such as ferrous scraps [5]. In contrast, Belgian recyclers also looked into the potential of recycling non-metallic materials such as plastics to achieve a higher recycled mass fraction. Although plastic recycling is not as lucrative as metal recycling, there is still great potential value for secondary plastic production. Moreover, it provides environmental benefits and allows further reduction of waste being produced for disposal [23].

The strict recycling targets and scarcity of available landfill space in Belgium have further encouraged minimal ELV waste disposal due to high landfill costs. This is in line with the ambition of preventing waste to landfill while stressing reuse, recycling, and waste incineration in accordance to Lansink's ladder [24]. Therefore, the implementation of advanced post-shredder technologies is continuously progressing since the associated recycling costs are still below the disposal cost. On the contrary, landfilling of untreated ASR is cost effective for vehicle recyclers in Australia due to the large availability of landfill space. Moreover, the recyclers are not held financially accountable for the environmental and societal impact. The economic incentives play a major role in the current ELV recycling; however, the implementation of strict legislation in Belgium is crucial to adjust the current ELV recycling procedures through the influence on recycling costs, including fines.

4. Environmental Impact Assessment Method

4.1. Project Scope and System Boundary

This paper evaluates the environmental impacts of different ELV recycling strategies based on a recycling facility in Australia and Belgium. The analysis only considers the material recycling and recovery efficiencies after the depollution process, as highlighted in Figure 1. Although part reuse provides better environmental gain, it was not considered due to the complexity of gathering the data. While the different non-ferrous (NF) materials were not further recovered in the Australian recycling facility, the NF mixtures were exported to developing countries and assumed to be further recovered via hand-sorting.

4.2. Functional Unit

The functional unit for this study was the recycling of a depolluted car hulk with an average mass of 852kg. The material composition of an average depolluted car hulk was based on the information provided by the Belgian recycling facility, as shown in Figure 2.

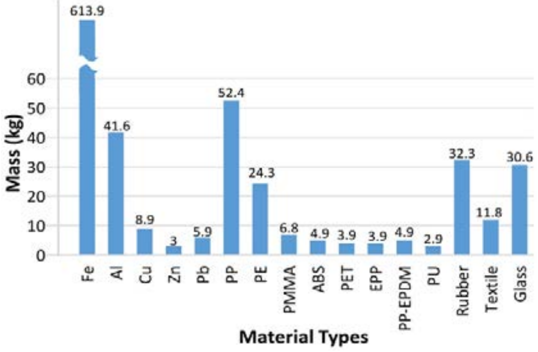

Figure 2: Average material composition of a depolluted car hulk.

4.3. Life Cycle Inventory (LCI)

The ELV recycling and recovery efficiencies for each material were gathered from two facilities, one located in Australia and one in Belgium. Data provided from the Belgian recycling facility were calculated based on the average ELV material flows in the plant. The information was then used to infer the material efficiencies for an average depolluted car hulk. For the Australian recycling facility, material recycling efficiencies were measured for car doors, and extrapolated to represent the entire vehicle. Car doors consist of variety of material mixtures that underwent different manufacturing processes commonly used for producing vehicles. Certain materials such as lead (Pb) and different types of plastics, that are present in an average depolluted car hulk, were not observed in a car door. Despite of that, these materials were not recovered in the Australian recycling facility. It was assumed that the process and material recycling efficiencies for the car doors can be reflected for generic ELV recycling. Samples taken from the NF stream were further hand-sorted to different types of NF metals to estimate their respective recycling efficiencies.

GaBi Professional v6.115 was used to model material scrap recycling, energy credit, and material landfilling. Down-cycling impact was included for metal recycling through the use of value-corrected substitution available in the database. This approach accounted for the avoidance of producing a mix of virgin and recyclate based on the scrap-to-virgin price ratio. The degree of down-cycling was determined from the relationship between material quality and scrap price to come up with the value correction factors [25]. Non-metal recycling only considered the avoidance of virgin material production due to the lack of data. Material recycling, energy recovery and landfill scenarios for different material types were based on the GaBi Professional database. Incineration was used to represent the energy recovery process.

4.4. Life Cycle Impact Assessment (LCIA)

The environmental impacts of ELV recycling were calculated based on the midpoint categories recommended by the International Reference Life Cycle Data System (ILCD) [26]. These recommendations were based on evaluated impact categories in existing LCIA models. The generated midpoint results were then normalised to person-equivalent to allow comparison between the overall environmental impact in both recycling facilities. The person-equivalent unit refers to the impacts caused by one person during one year in the European context.

5. Differences in Recycling Processes and Material Flows

The ELV recycling processes in the Australian and Belgian recycling facilities are shown in Figure 3 and Figure 4. Overall, the Belgian recycling facility has a better recycling efficiency when compared to the Australian recycling plant, as seen in Table 2. The recycling and recovery efficiency in Belgium is 94.33% whereas in Australia, the recycling efficiency is 71.61%. The efficiency of the Belgian recycling facility is lower than the set target of 95% because the reused ELV parts were not considered in this study. As detailed in Table 1, the difference in process efficiency is largely due to higher metal recycling in Belgium, with a minimum 91.76%. Fe has the highest recycling efficiency in both facilities; nevertheless, there is potential for the Australian recycling facility to further increase it by 3%. The recycling efficiencies for NF materials in the Australian facility are comparatively low, ranging from 4.11% to 41.9%. This is due to reduced focus on NF retrieval, and a reliance on eddy current separation. Stainless steel recycling was not included due to the lack of data.

ASR that would be landfilled in the Australian recycling facility undergo further treatment processes in the Belgian recycling facility. The post-shredder treatment utilises density separation to further segregate the variety of non-metallic materials and heavy metals. Plastic recycling is the focus in this process, and the recovered plastics are further sorted to different plastic types to improve purity and thus increase the value of secondary plastics. However, the recycling efficiencies varied vastly from one plastic type to another. Polypropylene (PP) and polyethylene (PE) have the highest recycling efficiency in the Belgian facility, both 89.5%, followed by Acrylonitrile Butadiene Styrene (ABS), which is about 83.4%. These plastic types are the most widely used in vehicle production. Conversely, other plastic types such as polymethyl methacrylate (PMMA), polyethylene terephthalate (PET), expanded polypropylene (EPP), polypropylene blended with ethylene propylene elastomer (PP-EPDM), and polyurethane (PU) each have a recycling efficiency below 0.7%.

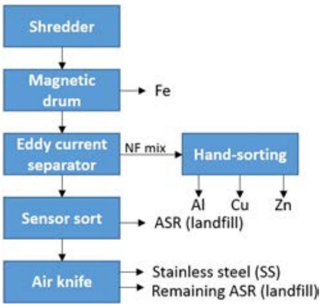

Figure 3: ELV recycling processes in the Australian recycling facility.

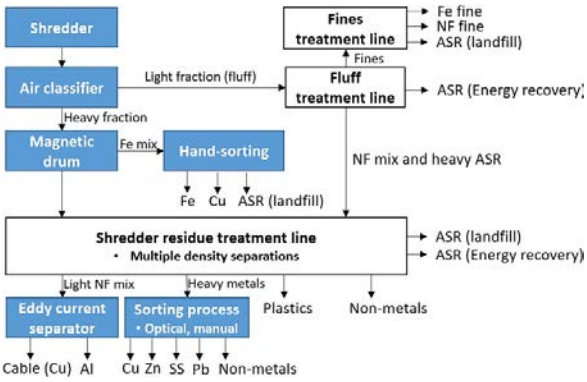

Figure 4: ELV recycling processes in the Belgian recycling facility.

Table 1: ELV recycling efficiencies in the Australian and Belgian recycling facility.

ELV Materials	Fe	Al	Cu	Zn	Pb	PP	PE	PMMA	ABS	PET	EPP	PP-EPDM	PU	Rubber	Textile	Glass
Australia	96.13	41.9	4.11	36.45	-	-	-	-	-	-	-	-	-	-	-	-
Belgium	99.97	97.52	91.76	98.24	97.08	89.50	89.50	0.32	83.36	0.40	0.58	0.32	0.69	1.94	0.83	79.40

Table 2: Post-shredder waste landfilled, recycling and recovery efficiencies for the Australian and Belgian recycling facilities.

Output Stream	Australia (%)	Belgium (%)
Material recycling	71.61	90.26
Energy recovery	-	4.07
ASR landfill	28.39	5.67

6. Environmental Impact Assessment Results

The material recycling in both facilities provides substantial environmental gain to compensate for the negative impact of energy recovery and landfilling. Although thermal energy recovered from waste incineration generates environmental offset, harmful gases are emitted. This has consequently caused the negative environmental effects of energy recovery. For the Australian scenario, landfill of unrecovered materials has reduced the potential environmental benefits significantly. This is particularly the case for metals that end up in the ASR fraction. The overall midpoint results based on the recycling and recovery of different materials are shown in Figure 5.

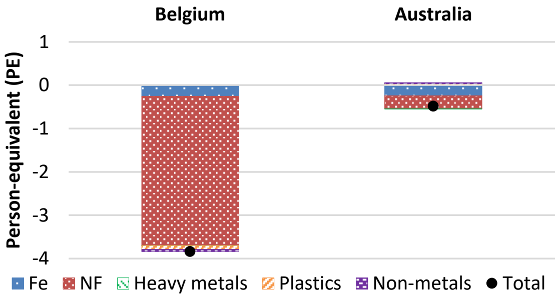

Figure 5: Overall midpoint results for both recycling facilities based on a processed car hulk (852kg).

There is still potential to further improve the environmental performance for both facilities through better material recycling efficiency. It is estimated that by improving the NF recycling efficiency to 92% for the Australian scenario, the environmental potential gains for NF recycling can increase by at least 11 times. The substantial increase in environmental benefits is due to the energy savings of secondary materials processing for Al and Cu in comparison to the extraction and production of primary resources. For instance, Al recycling requires 95% less energy than production from raw materials which subsequently reduces the greenhouse gas emissions. In Belgium, copper recycling can be further optimised to a higher recycling efficiency in comparison to other NF materials. The slightly lower recycling efficiency is due to the difficulty in retrieving copper used in electrical wiring and power motors. An improvement of 3% for the current copper recycling efficiency would provide an additional environmental gain of 3.1% based on the midpoint results.

ASR landfill and incineration contribute to the negative environmental impact for both facilities. The landfilling of plastics and other non-metal materials, such as rubber, textile and glass, has reduced the environmental gains for the Australian scenario. Although the negative impact can be neglected, these materials can provide further environmental benefits when recycled. For instance, the amount of plastic recycled from 500,000 units of ELV in 2010 could potentially avoid about 7.11E+07 kg CO2-equivalent. Based on the Belgian scenario, plastics and non-metal recycling can improve the environmental performance by 3.6%. There is potential to further maximise the plastic recycling for the Belgian scenario since plastic types with low recycling efficiency are either incinerated or landfilled.

7. Discussion/Analysis

Although the ELV recycling processes in Belgium produce high material recycling and recovery efficiencies, the number of ELVs treated within the country is low compared to Australia. In 2010, about 83% of deregistered vehicles were treated in Australia, whereas only approximately 50% of deregistered vehicles were treated in Belgium. The low number of ELVs treated in Belgium is largely due to the exportation of deregistered vehicles to other countries [27]. Consequently, the ELV treatment and recycling of exported vehicles will depend on the location of the last owner.

The vehicle's material composition has an impact on the overall material recycling efficiency. For older vehicles with high metallic content, Fe and NF recycling should be emphasised for better environmental performance. Although the recycling efficiencies for metals in the Belgian recycling facility are relatively high, the impurities present in the recovered streams need to be considered to account for a more realistic calculation of material recycling efficiency. Moreover, the types of impurities observed in different recovered material streams can have a large influence on the material quality of secondary production. The increasing complexity of multi-material vehicle designs will consequently cause the effect of cascade recycling that needs to be considered for future ELV recycling [28,29].

Increasing use of different plastic types in newer vehicle designs [5,30] will require necessary improvements for plastic recycling and reduced recovery. Proper reuse and recycling of plastics provide positive environmental gains through the avoidance of material loss and the extensive use of primary resources for plastic production. In this study, the environmental benefits from plastic recycling is not significant due to the small plastic content of the analysed car hulk. Nevertheless, plastic recycling will be crucial to lower the environmental impact of future ELV recycling due to the growth of the variety of plastics used in lightweight vehicles.

The midpoint results are limited to the inconsideration for the sensitivity of varying material input through the recycling processes, which can lead to the uncertainty of material output recycling efficiencies. There is potential to improve the uncertainty of material recycling and recovery efficiencies through the collection of multiple data to calculate the deviation from the average value.

8. Conclusions

One of the major similarities between both recycling facilities is the focus on recycling valuable materials for financial gain. The types of recycled materials are strongly influenced by the materials' market value to fully optimise the generated revenue. In this case study, both recycling facilities opted for the shredder-based recycling technology that has been proven to be cost effective for ELV recycling. The Australian recycling facility has a relatively high Fe recycling efficiency compared to other metals due to the high demand to provide enough stock for their affiliated steel mill company. In Belgium, high recycling efficiencies for different metals are achieved to maximise profit.

The enactment of strict ELV legislation has a significant impact on the ELV environmental performance. In countries where vehicle recyclers are governed by strict regulations, they are constantly pressured to improve the amount of materials recovered. Therefore, the ELV recycling performance in Belgium has a better environmental result compared to Australia, by 7.9 times. The large difference in the overall environmental impact is mainly caused by the strict legislative requirements and the use of advanced post-shredder technologies to further recover different NF and plastic materials.

Material recycling efficiency is strongly related to the adoption of recycling technologies. In the Belgian recycling facility, post-shredder technologies are used to retrieve the different types of plastics due to the market potential of secondary plastic production. The increasing use of plastic materials in current vehicle designs envisages great market opportunities for plastic recycling when these vehicles reach their EoL. Moreover, advancements in post-shredder technologies will ensure the recyclability of future vehicles abides by the strict recycling targets. In Australia, the material recycling efficiency is relatively low due to inefficient recycling processes. Although the adoption of more advanced recycling technologies can further improve the material recycling efficiency, revenues from recovered materials and strict policy play a significant role to actualise the transition.

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References

[1] Sakai S, Yoshida H, Hiratsuka J, Vandecasteele C, Kohlmeyer R, Rotter VS, et al. An international comparative study of end-of-life vehicle (ELV) recycling systems. J Mater Cycles Waste Manag 2014;16:1–20.

[2] ANFAC. European Motor Vehicle Parc 2010. 2012.

[3] Febelauto. Figures and results n.d. http://www.febelauto.be/userfiles/file/collection%20and%20recovery%20results%202010-2014.pdf.

[4] McNamara N. Vehicle Recycling and Sustainability. Melbourne, Australia: International Specialised Skills Institute Inc; 2009.

[5] Soo VK, Compston P, Doolan M. Is the Australian Automotive Recycling Industry Heading towards a Global Circular Economy? – A Case Study on Vehicle Doors. Procedia CIRP 2016;48:10–5.

[6] The Parliament of the Commonwealth of Australia. Product Stewardship Act 2011. 2011.

[7] ABRI. Australian Battery Recycling Initiative n.d. http://www.batteryrecycling.org.au/home (accessed August 16, 2016).

[8] Department of the Environment and Energy, Australia. Product Stewardship for Oil (PSO) Program n.d. https://www.environment.gov.au/protection/used-oil-recycling/product-stewardship-oil-program (accessed August 16, 2016).

[9] Department of the Environment and Energy, Australia. Product stewardship for end of life tyres n.d. https://www.environment.gov.au/protection/national-waste-policy/tyres (accessed August 16, 2016).

[10] Department of the Environment, Australia. National Waste Policy: Less Waste, More Resources. Natl Waste Policy Fact Sheet 2009. http://www.environment.gov.au/system/files/pages/94aa70c5-6681-44c6-8d83-77606d1d6afe/files/fs-national-waste-policy.pdf (accessed April 6, 2016).

[11] Vermeulen I, Van Caneghem J, Block C, Baeyens J, Vandecasteele C. Automotive shredder residue (ASR): Reviewing its production from end-of-life vehicles (ELVs) and its recycling, energy or chemicals' valorisation. J Hazard Mater 2011;190:8–27.

[12] Wright Corporate Strategy Pty Limited. Review of the Application of Landfill Standards. 2010.

[13] Department of Environment and Heritage. Environmental Impact of End-of-Life Vehicles: An Information Paper. Australia: 2002.

[14] Dawkins E, Allan P. Landfill Ban Investigation. Hyder Consulting Pty Ltd; n.d.

[15] Kanari N, Pineau J-L, Shallari S. End-of-life vehicle recycling in the european union. JOM 2003;55:15–9.

[16] E. U. Directive. 53/EC of the European Parliament and of the Council of 18 September 2000 on end-of life vehicles. Off J Eur Union Ser 2000:34–42.

[17] Smink CK. Vehicle recycling regulations: lessons from Denmark. J Clean Prod 2007;15:1135–46.

[18] Sander K, Schilling S, Zangl S, Lohse J. Rules on compliance with Article 7.2 of Directive 2000/53/EC. ÖKOPOL Inst Ecol Polit GmbH Rep Compil Dir Gen Environ Nucl Saf Civ Prot Comm Eur Communities Ger 2002.

[19] Febelauto n.d. http://www.febelauto.be/en/ (accessed August 17, 2016).

[20] Gerrard J, Kandlikar M. Is European end-of-life vehicle legislation living up to expectations? Assessing the impact of the ELV Directive on "green" innovation and vehicle recovery. J Clean Prod 2007;15:17–27.

[21] Mazzanti M, Zoboli R. Economic instruments and induced innovation: The European policies on end-of-life vehicles. Ecol Econ 2006;58:318–37.

[22] Bellmann K, Khare A. Economic issues in recycling end-of-life vehicles. Technovation 2000;20:677–90.

[23] Inghels D, Dullaert W, Raa B, Walther G. Influence of composition, amount and life span of passenger cars on end-of-life vehicles waste in Belgium: A system dynamics approach. Transp Res Part Policy Pract 2016;91:80–104.

[24] Wolsink M. Contested environmental policy infrastructure: Socio-political acceptance of renewable energy, water, and waste facilities. Environ Impact Assess Rev 2010;30:302–11.

[25] Koffler C, Florin J. Tackling the Downcycling Issue—A Revised Approach to Value-Corrected Substitution in Life Cycle Assessment of Aluminum (VCS 2.0). Sustainability 2013;5:4546–60.

[26] European Commission, Joint Research Centre, Institute for Environment and Sustainability. ILCD Handbook - General guide for Life Cycle Assessment-Detailed guidance 2010.

[27] European Parliament. End of life vehicles: Legal aspects, national practices and recommendations for future successful approach. 2010.

[28] Paraskevas D, Kellens K, Dewulf W, Duflou JR. Environmental modelling of aluminium recycling: a Life Cycle Assessment tool for sustainable metal management. J Clean Prod 2015;105:357–70.

[29] Soo VK, Compston P, Doolan M. Interaction between New Car Design and Recycling Impact on Life Cycle Assessment. Procedia CIRP 2015;29:426–31.

[30] U.S Department of Energy. Use of Lightweight Materials is on the Rise. Veh Technol Off 2013.https://www1.eere.energy.gov/vehiclesandfuels/facts/2013_fotw786.html (accessed February 24, 2014).