Roja Modaresi

Dynamics of aluminum use in the global passenger car system

Challenges and solutions of recycling and material substitution

Thesis for the degree of Philosophiae Doctor

Trondheim, May 2015

Norwegian University of Science and Technology
Faculty of Engineering Science and Technology
Department of Hydraulic and Environmental Engineering
Industrial Ecology (IndEcol) Programme

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Abstract

This thesis analyzes the relationship between the design of vehicles, end-of-life vehicle (ELV) management, and global material production using aluminum as an example. Vehicle manufacturing, material industries and ELV management face different challenges. An important challenge for vehicle manufacturers is the design of lightweight vehicles to reduce energy use and greenhouse gas (GHG) emissions in the use phase for which an increased use of aluminum of different alloys is an attractive option. The aluminum industry has an interest in reducing energy consumption and GHG emissions, which can be accomplished effectively through recycling. ELV management must be improved to enable the first two systems to use aluminum scrap in a sustainable manner. Today, the sorting of different alloys is limited. As a result of having mixed scrap at the ELV phase and limited opportunities for aluminum refining, there may be a future scrap surplus that cannot be absorbed by the aluminum-recycling sink, which is passenger cars. These three sectors are connected through material flows, and a change in one of the sectors can severely affect the others' options for reaching their goals.

This thesis addresses the following questions: 1) How are the dynamics of the global vehicle stock changing the boundary condition for aluminum recycling? 2) What are the most effective interventions to minimize a future aluminum scrap surplus? 3) What are the options for material substitution in vehicles to reduce direct and indirect GHG emissions over time?

To answer these questions, a system approach is employed to analyze how these three sectors are linked and to explore options for all sectors to reach their objectives in the long term. This thesis employs global bottom-up stock-driven models of the aluminum cycle. A basic model was used to identify the scrap surplus problem. A refined model with segments, components and alloys resolution combined with a source-sink diagram was used to evaluate different solution options. In addition, a global dynamic fleet-recycling MFA model was developed to simulate the future impacts of material substitutions of conventional steel with high-strength steel (HSS) and aluminum on material cycles, energy use and GHG emissions related to the global passenger vehicle fleet.

The main findings in this thesis are: i) a continuation of the current practice of cascadic use would eventually result in a scrap surplus because this practice depends on the continuous and fast growth of the secondary casting stock in the global vehicle fleet, a condition that is unlikely to be met. Model simulation indicated a non-recyclable scrap surplus by approximately 2018±5 if no alloy sorting is introduced. The surplus is potentially substantial and could grow to reach a level of 0.4–2 kg/cap/yr by 2050,

thereby significantly reducing the option of the aluminum industry to reduce its energy consumption through recycling. ii) Drastic changes in ELV management practices are necessary to make use of the growing scrap flow in the future, including further dismantling and efficient component-to-component recycling, alloy sorting of mixed shredded scrap, and designing recycling-friendly alloys that function as alternative sinks for aluminum scrap. iii) Light-weighting has the potential to substantially reduce global emissions of vehicles (9-18 gigatons cumulative CO_2-eq. between 2010 and 2050). In the medium term (5-15 years), global emissions reductions from substituting standard steel with aluminum are similar to those achievable by HSS; however, over a longer term (after 15-20 years), substitution with aluminum can reduce total emissions more effectively, provided that the wrought aluminum will be recycled back into automotive wrought aluminum.

The environmental consequences of products in general and passenger cars in particular have led to an increasing awareness of the dependencies between the shaping of vehicles and the shaping of the environment. Governments and intergovernmental bodies have formulated quality goals for the environment, such as the 2-degree target, and have introduced emissions standards, thereby extending the responsibility of automobile manufacturers to the use phase. On the materials side, legislation has been introduced to extend producer responsibility, mainly with the goal of avoiding toxic substances and reducing the amount of waste, as is noted in different end-of-life vehicle (ELV) legislation and directives. The current ELV directives do not sufficiently address the management of material systems as a whole or quality issues related to material recovery. To harmonize ELV management with goals for the global aluminum cycle and its impacts for the environment, it is essential to understand how the above-mentioned systems interact.

Acknowledgments

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Asker, February 2015

Roja Modaresi

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List of figures

List of appended papers and the author's contributions to them

All papers are published and co-authored. My contribution is indicated for each one.

Paper no. 1: (Published)
The Role of Automobiles for the Future of Aluminum Recycling. Modaresi, R.; Müller, D. B., Environ. Sci. Technol., 2012, 46 (16), pp 8587–8594. DOI: 10.1021/es300648w
Contribution: Part of the research design, data collection, model design and development, analysis, visualization, and writing.

Paper no. 2: (Published)
Aluminium Recycling – Raw Material Supply from a Volume and Quality Constraint System. Rombach, G.; Modaresi, R.; Müller, D. B., World of Metallurgy – ERZMETALL 2012, 65 (3).
Contribution: Data collection and visualization for the section related to automotive casting as a bottleneck for recycling.

Paper no. 3: (Published)
Component- and alloy-specific modeling for evaluating aluminum recycling strategies for vehicles. Modaresi, R.; Løvik, Amund N.; Müller, D. B., The Minerals, Metals & Materials Society (TMS), JOM, 2014, 66 (11) pp 2262-2271. DOI10.1007/s11837-014-0900-8
Contribution: Research design, data collection, model design and development, analysis, visualization (except for Figure 1), and writing.

Paper no. 4: (Published)
Long-Term Strategies for Increased Recycling of Automotive Aluminum and Its Alloying Elements. Løvik, Amund N.; Modaresi, R.; Müller, D. B., Environ. Sci. Technol., 2014, 48 (8), pp 4257–4265. DOI: 10.1021/es405604g
Contribution: Part of the research design, part of the data collection and refining.

Paper no. 5: (Published)
Global carbon benefits of material substitution in passenger cars until 2050 and the impact on the steel and aluminum industries. Modaresi, R.; Pauliuk, S; Løvik, Amund N.; Müller, D. B., Environ. Sci. Technol., 2014, 48 (18), pp 10776–10784. DOI: 10.1021/es502930w
Contribution: Research design, parts of the data collection, parts of the model design and writing.

1 Introduction

1.1 Summary of Introduction

Demand for material services has increased sharply over the past decades and is expected to increase further over the next decades as a result of industrial development and population growth (IEA, 2010, Allwood and Cullen, 2012, Krausmann et al., 2009). The consequences are higher materials and energy demand, as well as environmental challenges, such as resource depletion, water stress, land stress, and climate change.

The Intergovernmental Panel on Climate Change (IPCC) found that most of the observed increase in the global average temperature in recent decades is likely a result of anthropogenic GHG concentrations. The 5th Assessment report of the IPCC concludes that to limit the global average temperature increase to 2-2.4°C, a limit to avoid the dangerous effects of climate change, a drastic reduction in GHG emissions is vital: *'Scenarios reaching atmospheric concentration levels of about 450 ppm CO2eq by 2100 (consistent with a likely chance to keep temperature change below 2°C relative to pre-industrial levels) include substantial cuts in anthropogenic GHG emissions by mid-century through large-scale changes in energy systems and potentially land use.'* (IPCC, 2014b).

Figure 1 shows the direct and indirect GHG emissions across sectors in the baseline scenarios. The transport sector produced 6.7 $GtCO_2$ of direct GHG emissions in 2010 (Sims R., 2014, IPCC, 2014b) and was thus responsible for nearly 23% of global energy-related anthropogenic carbon emissions (Allwood and Cullen, 2012, Sims R., 2014). Road vehicles account for more than three-quarters of emissions from the transport sector, and the use of passenger cars accounted for nearly 75% of all road vehicles (OICA, Allwood and Cullen, 2012, Sims R., 2014, IPCC, 2014b). Therefore, direct emissions from passenger cars account for approximately 14% of total energy-related global anthropogenic carbon emissions. Not included in these emissions are the emissions from the production of passenger cars and road infrastructure, including material production, which are allocated to industry emissions.

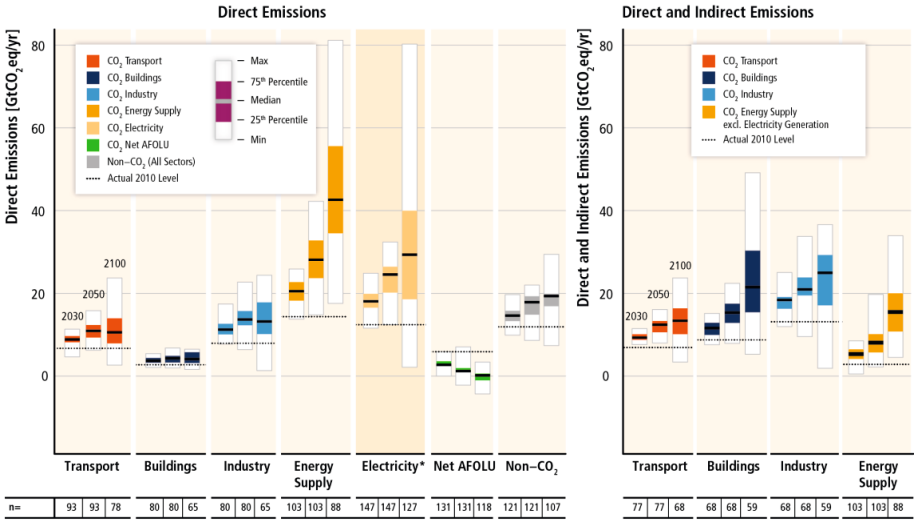


Figure 1 Direct (left panel) and direct and indirect emissions (right panel) of CO_2 and non-CO_2 GHGs across sectors in baseline scenarios. Non-CO_2 GHGs are converted to CO_2-equivalents based on Global Warming Potentials with a 100-year time horizon from the IPCC Second Assessment Report (SAR). In the case of indirect emissions, only electricity generation emissions are allocated from energy supplies to end-use sectors. In the left panel, electricity sector emissions are shown (Electricity*) in addition to energy supply sector emissions, of which they are part, to illustrate their large role on the energy supply side. (IPCC, 2014a)

To quantify the total emissions related to passenger cars and to explore options for their reduction, it is essential to regard passenger cars as a part of a larger system that includes three interlinked sub-systems: 1) vehicle and material production (from an emissions perspective dominated by material production), 2) Use phase or vehicle fleets, and 3) end-of-life vehicle management (see Figure 2).

The aim of this thesis is to analyze the socio-economic metabolism of this system with a focus on aluminum as a case study of a key material in vehicles. Models of the socio-economic metabolism can be defined as the set of processes that are connected by material and energy flows (Ayres and Simonis, 1994, Baccini and Brunner, 1991, Fischer-Kowalski et al., 2011, Fischer-Kowalski, 2011). In-use stocks are important elements of the socio-economic metabolism in material and substance flow analysis (MFA and SFA) (Müller et al., 2006, Müller, 2006, van der Voet et al., 2002, Pauliuk and Müller, 2013). In-use stocks, such as buildings, cars, and infrastructure, provide services to society and fulfill the need for major human activities, such as residing, working, transportation, and communication (Müller et al., 2010, Pauliuk and Müller, 2013, Baccini and Brunner, 1991).

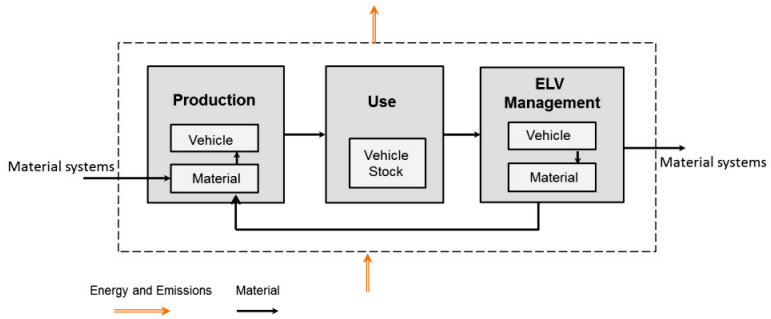


Figure 2 Generic material flow model from production to end-of-life

Following is a summary of key points for the global vehicles system, the global material system and the end-of-life management system and the main interlinks with the two other systems:

- The global vehicle stock has been growing remarkably as a result of population growth and lifestyle changes (IEA, 2012, Joyce Dargay, 2007, Bandivadekar et al., 2008). This has resulted in increasing material demand, scrap availability, and direct (use phase) and indirect emissions (production phases). The definitions for direct and indirect emissions are used differently in this study from the IPCC definition. We include material production emissions in addition to fuel production emissions for the indirect emissions.
- There is a strong correlation between car weight and fuel consumption. Car weight has risen in recent decades mainly as a result of the increased use of safety and comfort features. Finding a balance between weight reduction, safety and comfort is a challenge that can be resolved with light-weighting strategies, including material substitution, vehicle redesign, or a shift to smaller cars (Cheah, 2010, Kim et al., 2010a, Bandivadekar et al., 2008). All light-weighting strategies have an impact on material use and thus on end-of-life management.
- One of the largest and fastest growing reservoirs of aluminum in use resides in automobiles (EAA 2008). Aluminum use in new passenger cars has grown fivefold in the last three decades (Ducker 2009). The main penetration occurred with castings (used in engines, among other parts), whereas the major future growth potential is expected in wrought aluminum, used mainly in components for the body-in-white (BIW). This has significant implications for the recovery of aluminums in end-of-life vehicle management (and the recycling of aluminum in the automotive industry) because automobile cast aluminum is currently the only relevant sink for aluminum scrap from the automotive sector or other sectors.
- Steel is currently the most widely used material within the automotive industry. Due to the need for light-weighting strategies, high-strength steel (HSS) and

other light-materials, such as aluminum, started to be substituted for conventional steel in the last few decades.

- Efficient ELV management is becoming essential due to the increasing number of ELVs and the high market value for scrap from the vehicles. Although most national and regional legislation set specific goals to reach a certain overall recycling rate, goals for the quality of the recovered materials, for example, the purity of different alloys and the recovery of critical metals, which are used in small amounts, are largely ignored. In the case of aluminum, different alloy types are often recovered as one mixed aluminum scrap fraction. Due to refining difficulties, the aluminum scrap is blended and recycled to alloys that can accept a higher amount of alloying elements. This alloy cascade can potentially limit the use of this mixed scrap in the future.

This thesis tests the following hypothesis: a continuation of the current aluminum recycling practice will eventually lead to the formation of large amounts of highly alloyed material, which may no longer find an application in the automotive market. Such a resource loss would have implications for energy use because the surplus scrap could not be used to replace aluminum from primary production, resulting in an unusable energy savings potential. Suitable policy options must be identified to avoid or delay the scrap surplus problem.

In this thesis, the following questions are addressed: 1) How are the dynamics of the global vehicle stock changing the boundary conditions for aluminum recycling? 2) What are the most effective interventions to minimize a future aluminum scrap surplus? 3) What material substitution options for vehicles exist to reduce direct and indirect GHG emissions over time?

Five papers have been developed to answer the proposed research questions. These papers constitute the core of this thesis which is shown in Figure 3.

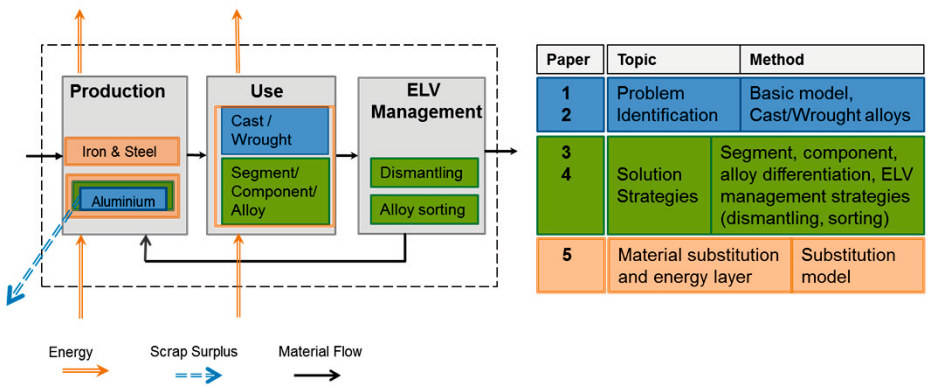


Figure 3 Structure of the thesis and organization of the appended papers

In the introduction section, trends in private transportation, material use, and ELV management are described in chapters *1.2* to *1.4*. These trends provide a basis for an elaboration of potential challenges and solutions for aluminum recycling (chapter *1.5*), as well as the motivation and research questions (chapter *1.6*).

1.2 Trends in private transportation

1.2.1 Global vehicle stock growth due to population and car ownership growth

Worldwide transportation studies confirm that the vehicle stock has been increasing over time due to population growth and lifestyle changes (IEA, 2010, Bandivadekar et al., 2008, Joyce Dargay, 2007, IEA, 2012). According to the IPCC fourth assessment report (IPCC, 2007), the world auto fleet has grown with remarkable speed – between 1950 and 2000, the number of vehicles increased from approximately 50 million vehicles to 600 million vehicles, an increase that is five times higher than the growth in population over the same period. This fact can also be observed in Figures 4-6.

It is expected that the global vehicle fleet will surpass 2 billion units by 2030 (Joyce Dargay, 2007, IEA, 2012). Dargay (2007) projected the global vehicle fleet based on pooled time-series data from 1960-2002 and cross section data for 45 of the most populated countries, assuming different saturation levels for the studied countries by accounting for the proportion of the urban population and population density, economic development and per capita income. The result of the study shows that the percentage of the world's vehicles owned in non-OECD countries was 24% in 2002 and is expected to increase to 56% by 2030. In particular, China's vehicle stock is expected to increase by a factor of 20 in the period from 2002 to 2030 (Joyce Dargay, 2007).

This remarkable growth in vehicle stock has significant implications for material demand, scrap availability, and direct and indirect emissions, as well as scrap to be handled by the ELV management system.

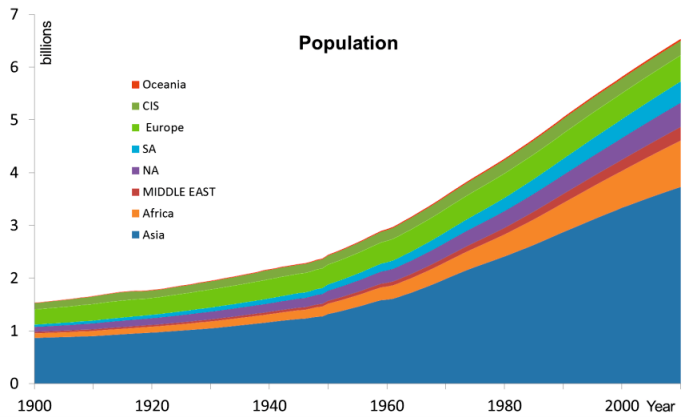


Figure 4 Population in different world regions. (UN, 2003, UN, 2010).

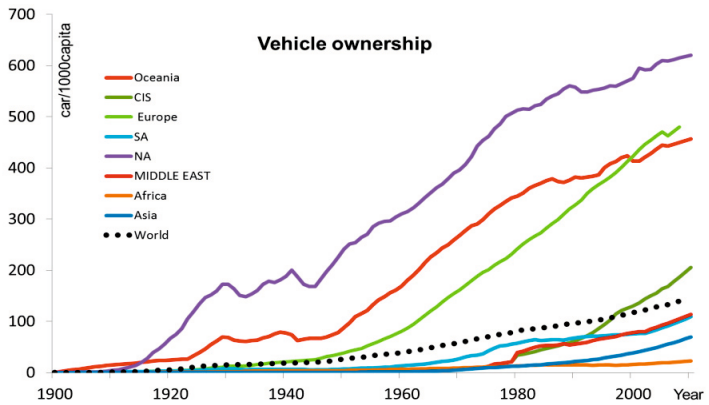


Figure 5 Passenger car ownership in different world regions (TheWorldBank, 2013, Joyce Dargay, 2007, Mitchell, 2007b, Mitchell, 2007c, Mitchell, 2007d, Mitchell, 2007a).

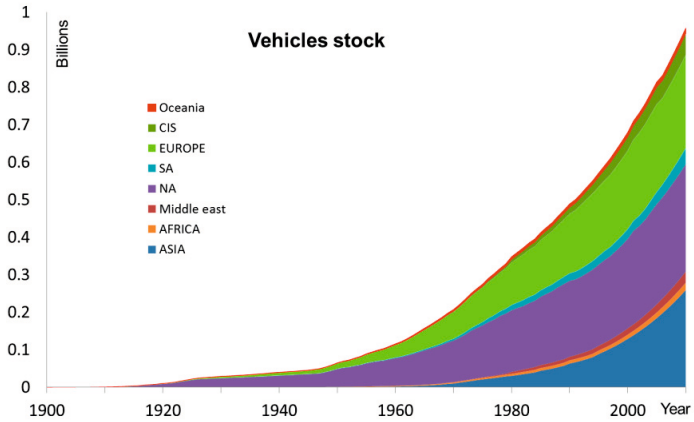


Figure 6 Passenger car stock in different world regions (calculated from data shown in Figures 4 and 5).

1.2.2 Segments

Although trends in some developed countries, such as those in North America, exhibit a tendency toward larger cars (Cheah, 2010), global vehicle production has a tendency toward smaller car segments in recent years (see Figure 7). This may be explained by the use of smaller cars in developing countries and emerging economies, such as India and China.

There are several different methods of car classification around the world. Generally, the boundaries between car segments are defined by factors, such as size and weight. The most common classification, according to European Commission regulation (EEC) No 4064/89, is: A-segment (mini cars, such as Ford Ka, Smart), B-segment (small cars, such as Ford Fiesta), C-segment (medium cars, such as Ford Focus, Honda Civic), D-segment (large cars, such as Audi A4, BMW 3 series), E-segment (executive cars, such as Ford Taurus) and F-segment (full-frame and luxury cars, such as Audi A8, BMW 7 series). Sport coupes, multi-purpose, and sport utility (including off-roads), which are categorized as S, M, and J-segments, respectively, do not have a significant global market share and are thus not considered in Figure 7.

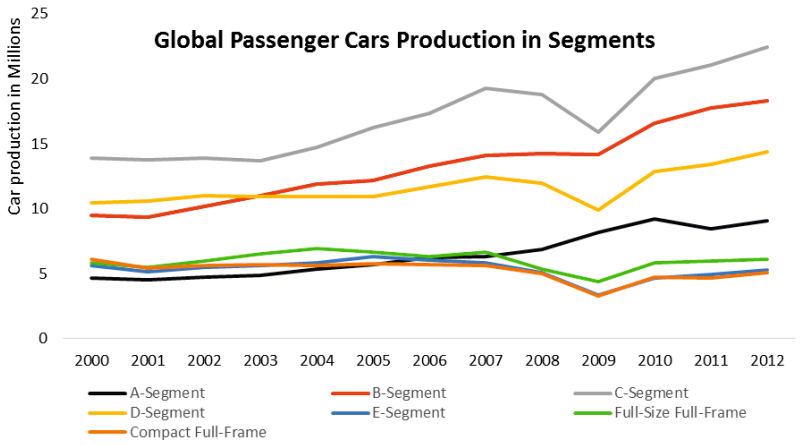


Figure 7 Global passenger car production with segments details (IHS, 2010).

1.2.3 GHG emissions

Human activities are tightly linked to material production and consumption. The transport sector contributes significantly to global energy-related anthropogenic carbon emissions, and passenger cars account for approximately 14% of total energy-related global anthropogenic carbon emissions (OICA, Allwood and Cullen, 2012, Sims R., 2014, IPCC, 2014b).

Figure 8 shows the key contributors to global anthropogenic CO_2 emissions arising from energy production and industrial processes. In addition to direct emissions from the use phase, passenger cars also contribute to initial emissions from the upstream production of materials. The pie chart shown on the right in Figure 8 shows the contribution of the industrial carbon emissions, which include 36% of total global human-made carbon emissions. This pie chart shows the contributions from the production of different materials in global industrial activities. Steel and aluminum contribute 25% and 3%, respectively, to industrial carbon emissions, representing 9% and 1% of global anthropogenic carbon emissions.

Various materials are used in car production; inter alia, steel and aluminum play a significant role in car production and global material GHG emissions. Material selection and specifications for vehicles are complex processes governed by a broad set of requirements, including functional performance and physical/chemical properties, structural integrity, safety, durability, aesthetics, material and fabrication costs, and recyclability (Keoleian and Sullivan, 2012).

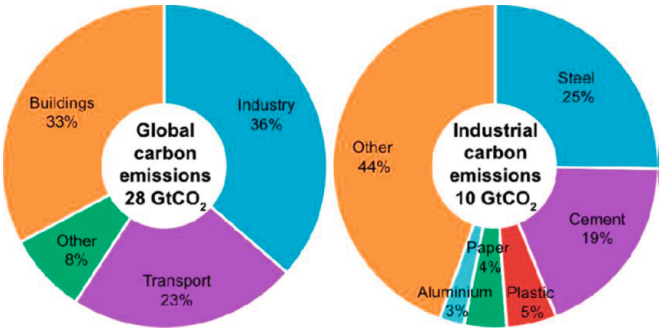


Figure 8 Global anthropogenic CO_2 emissions related to energy and industrial processes for the year 2006 (Allwood and Cullen, 2012).

1.2.4 Correlation between vehicle weight and fuel consumption

There is a strong correlation between car weight and fuel consumption. A 10% reduction in car weight typically reduces the fuel consumption by approximately 3-7% (Allwood and Cullen, 2012, Cheah, 2010, Bandivadekar et al., 2008, Martin Johannaber, 2007, Kim and Wallington, 2013). The first oil crisis in the early 1970s and the adaptation of fuel economy standards in North America made the automotive industry adopt light weighting measures to improve fuel economy (Horvath, 2010). The average weight of North American vehicles declined by approximately 20% in the period from 1976 to 1986 (Ducker, 2011b). However, vehicles have become heavier since then (Figure 9), largely due to added safety and comfort equipment as well as

customer demand for larger cars (EAA, 2006, EAA, 2012, Pinto, 2009). To reach the desired performance level, the weight of components, such as the engine, transmission, and brakes, had to be increased accordingly (EAA, 2012). This phenomenon is known as the 'weight spiral.' Today, one of the most demanding dilemmas for the automotive industry is to reduce overall vehicle weight while providing high levels of safety and comfort (EAA, 2012).

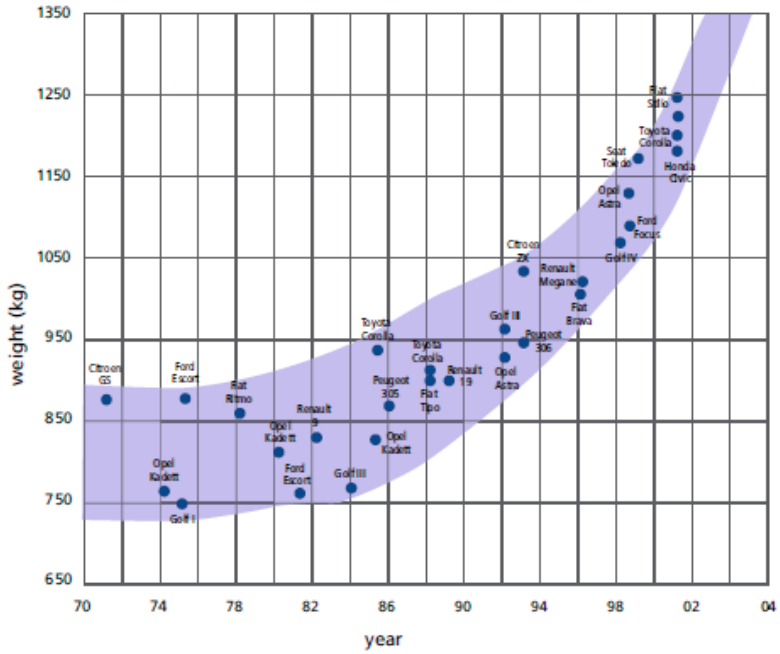


Figure 9 Evolution of weight in the compact class during the period 1970-2004 (EAA, 2006).

New regulations for direct emission reductions appear to be a strong incentive for extra light-weighting efforts among car manufacturers. EU regulations, for example, set emission targets for the average fleet of new cars entering the market to 130 grams of CO_2-eq. per kilometer beginning in 2015 (ICCT, 2014), and the US Corporate Average Fuel Economy (CAFÉ) regulation set a target of 102-133 grams of CO2-eq. per kilometer for the average fleet from 2025 (An et al., 2011).

To invert the weight spiral, a reduction in vehicle mass is essential, in addition to other technological developments for fuel efficiency. A reduction in vehicle weight can be achieved by different strategies that may be combined, including (1) material substitution, (2) vehicle redesign, and (3) vehicle downsizing (Cheah, 2010). Material substitution strategies involve the use of light-weight materials that can fulfill several criteria, such as economic viability, weight saving potential, and environmental and safety factors (Ghassemieh, 2011, Allen et al., 2007), in addition to physical properties, such as strength, stiffness and formability (Cheah, 2010, Kim and Wallington, 2013).

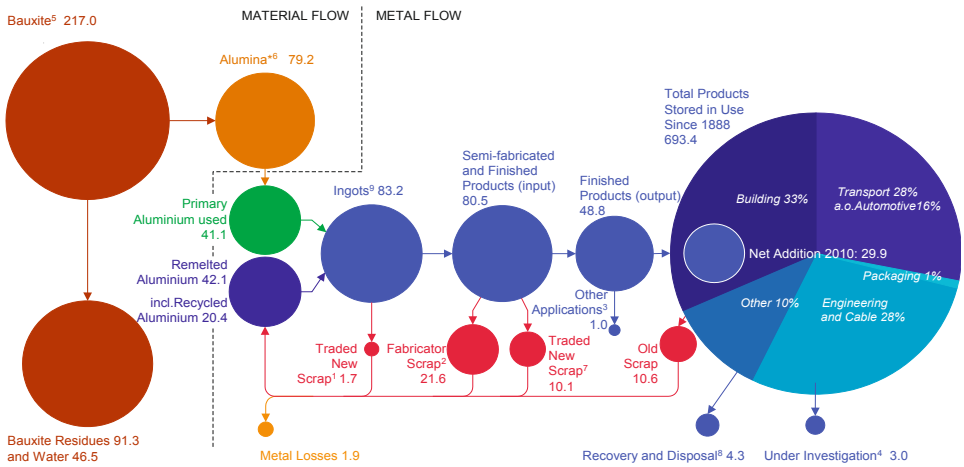
Because aluminum and high-strength steel (HSS) are the most cost-effective candidates for light-weighting in large-scale production, their use in car manufacturing is expected to increase in the future (Kim et al., 2010a, Ghassemieh, 2011, Cheah, 2010, Bandivadekar et al., 2008, Allwood and Cullen, 2012).

1.3 Trends in material use

1.3.1 Aluminum

Although it is the most abundant metal in the earth's crust (approximately 8% by weight), aluminum was identified as a metal only in 1808, and the first commercial aluminum production process, developed by Henri Sainte-Claire Deville, started in 1855 (Totten and MacKenzie, 2003). Today, aluminum is the second most used metal after iron and steel, and similar to steel, aluminum is often used in alloyed form to augment performance. Global aluminum primary production increased from 5 to 35 million tons per year between 1960 and 2005 (IAI, 2009). Figure 10 shows the global aluminum mass flow model (GARC, 2011) from bauxite extraction to the end-of-life for the year 2010.

The main raw material for aluminum production is bauxite, which is extracted from bauxite mines, then processed to alumina, which is used for aluminum production in an electrolytic process. The abundance of bauxite is relatively limited. Aluminum production results in large amounts of red mud, which needs special handling due to residual alkaline content. Currently, red mud is typically deposited near mining sites in sealed ponds from which excess water is returned to the bauxite mining process (EAA, 2008).



Values in millions of metric tonnes. Values might not add up due to rounding. *Change in stocks not shown
1 Aluminium in skimmings; 2 Scrap generated by foundries, rolling mills and extruders. Most is internal scrap and not taken into account in statistics; 3 Such as deoxidation aluminium (metal property is lost) 4 Area of current research to identify final aluminium destination (reuse, recycling, recovery or disposal); 5 Calculated based on IAI LCI report - update 2005. Includes, depending on the ore, between 30% and 50% alumina; 6 Calculated. Includes on a global average 52% aluminium; 7 Scrap generated during the production of finished products from semis; 8 Either incinerated with/without energy recovery, material recovery or disposal; 9 Estimated stock increase 980,000 tonnes.

Figure 10 Global aluminum mass flow model. The model was developed by IAI's Global Aluminium Recycling Committee (GARC) (GARC, 2011)**.**

Another concern regarding the primary production of aluminum is greenhouse gas (GHG) emissions and energy use. Overall, the production of primary aluminum accounts for 1% of world GHG emissions (Allwood and Cullen, 2012, McMillan et al., 2012). GHG emissions occur in all production stages, from bauxite mining to aluminum production, as shown in Figure 11 (Liu et al., 2013a). The figure shows that smelting and other primary-production-related processes (mining, refining and producing anodes for smelting) together are responsible for over 90% of total aluminum production emissions.

Between 1990 and 2005, the global average intensity of electricity consumed by primary aluminum smelters decreased from 16.5 to 15.6 kWh/kg (McMillan and Keoleian, 2009). Modern primary aluminum production facilities consume 13-14.1 kWh/kg of aluminum, which is approximately double the thermodynamic limit of 6.3 kWh/kg (IAI, 2009). Therefore, further improvements will become increasingly difficult to achieve.

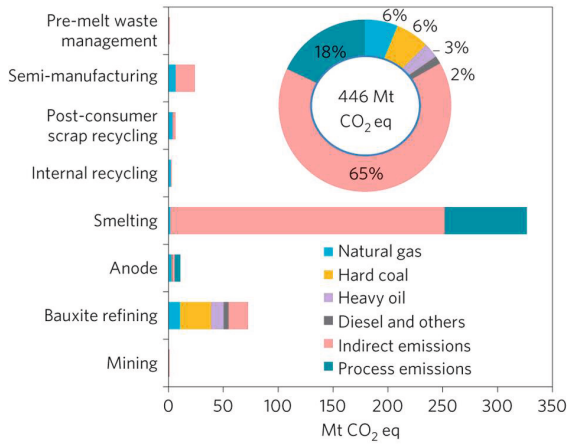


Figure 11 GHG emissions in all production stages of the global aluminum cycle in 2009 (Liu et al., 2013a).

In contrast, aluminum production from scrap requires approximately 20 times less energy (IAI, 2009). Aluminum, once produced, embodies large amounts of energy that can be saved if the end-of-life scrap is recovered for recycling. For this reason, aluminum in use can be regarded as an energy bank. Figure 12 shows that the ratio of secondary aluminum production from all types of scrap, including manufacturing and post-consumer scrap, grew from 17% in 1960 to 33% in 2006, and is expected to reach 40% by 2040 (IAI, 2009).

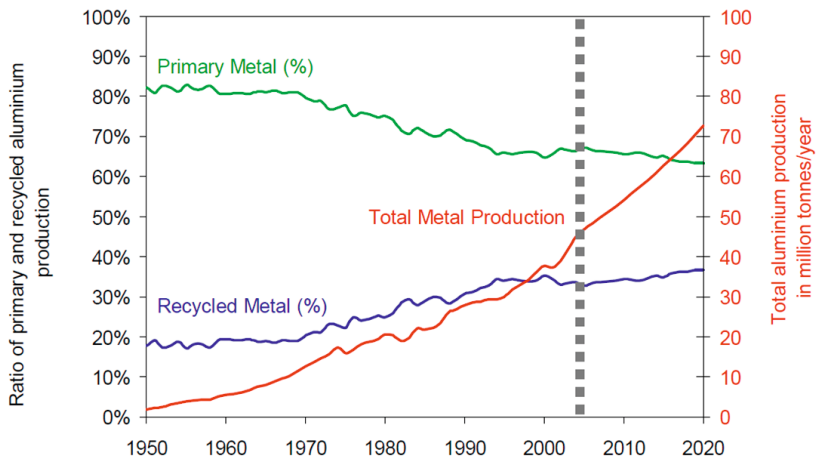


Figure 12 Share of recycled and primary aluminum (IAI, 2009).

The anthropogenic aluminum cycle has been quantified and analyzed using mass balances or quasi-stationary and dynamic MFA models. Chen and Graedel (2012) provide a review of anthropogenic element cycles with 26 different MFA studies of

aluminum across a range of geographical scales and lifecycle stages (Chen and Graedel, 2012).

Mass balances and quasi-stationary MFA models were used to characterize the aluminum cycle for a single year or selected years on a country level, inter alia, Italy (Amicarelli et al., 2004), the US (Plunkert, 2006), and China (Chen et al., 2010). Dynamic MFA models of the aluminum cycle in individual countries were introduced for Germany by Bever (1976) and later refined by Melo (1999), and for the US by Hatayama et al. (2009), Chen and Graedel (2012), Liu et al. (2011), and McMillan et al.(2010). The aforementioned studies considered historical consumption data and product lifetimes to calculate scrap generation. A study by Boin and Bertram (2005), conducted at the EU level, traced aluminum scrap flows coming from six sectors of building, transportation, beverage, foil, engineering and consumer durables.

The study of aluminum through MFA was further expanded to the global scale by the Global Aluminium Recycling Committee (GARC) (Bertram et al., 2009b, Martchek, 2006). Figure 10 shows the global aluminum mass flow model – GARC for the year 2010 (GARC, 2011).

Cullen and Allwood (2013) mapped the global flow of aluminum from liquid aluminum to end-use goods and illustrated the results in a Sankey diagram for the year 2007. The main focus of the study was to understand the material efficiency of the industry and recycling of post-consumer scrap by providing a detailed analysis of aluminum scrap flows and considering dilution with primary materials to reach the required quality. Predictions of global aluminum demand are typically calculated based on extrapolations of market growth assumptions (Rombach, 2002, Schwarz et al., 2001) or economic indicators, such as price or per-capita GDP (Luo and Soria, 2008, Menzie et al., 2010).

Liu et al. developed a dynamic material flow analysis model to simulate the future global aluminum cycle and emissions pathways and mitigation potentials (Figure 13). The model enables an integrated analysis of the material, energy and emissions nexus. It considers i) system feedbacks, which mean the scrap availability influences primary production, and ii) time lags, which mean the accumulation and replacement of in-use stocks is calculated based on the mass balance principle. Liu et al. calculated historic aluminum in-use stocks based on production, trade data, and product lifetime assumptions, and future aluminum based on the scenarios measured by the stock-driven model (Liu et al., 2013a). According to the study, the global aluminum in-use stock has reached approximately 90 kg per capita in 2009, with a range of 10-60 kg per capita for developing countries and 200-600 kg per capita for developed countries.

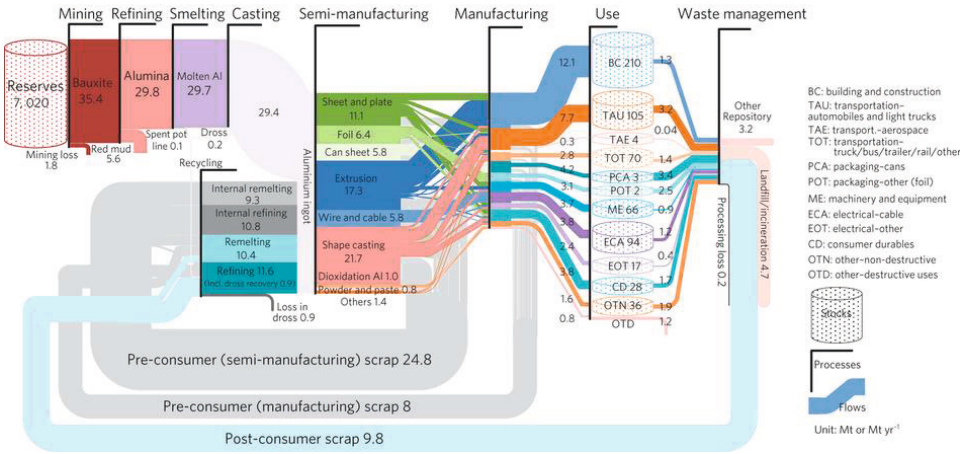


Figure 13 Global anthropogenic metallurgical aluminum cycle in 2009 (Liu et al., 2013a). The flows' widths are proportional to their magnitude. Building and construction (210 Mt), transportation (180 Mt, TAU, TAE, and TOT together) and electrical engineering (110 Mt, sum of ECA and EOT) constitute the largest components of the global aluminum in-use stock (636 Mt).

1.3.2 Steel

The global annual demand for steel is ca.1.4 billion tons (Gt) per year (Menzie et al., 2013). This high material demand is driven by the need to create and maintain the stock of steel products mainly used in construction, vehicles, industrial equipment, and metal products, such as packaging and appliances (Cullen et al., 2012). Steel production accounts for 25% of industrial carbon emissions and 9% of global anthropogenic energy- and process-related greenhouse gas emissions; therefore, climate change mitigation may represent a major constraint to future production growth (Allwood and Cullen, 2012). To develop roadmaps for emission reductions, information on trends in steel use, steel demand, and scrap availability is required. Figure 13 (Cullen et al., 2012) shows the global steel flow in 2008 from steelmaking, from intermediate products to end-use goods, and the complex interactions of the steel supply chain.

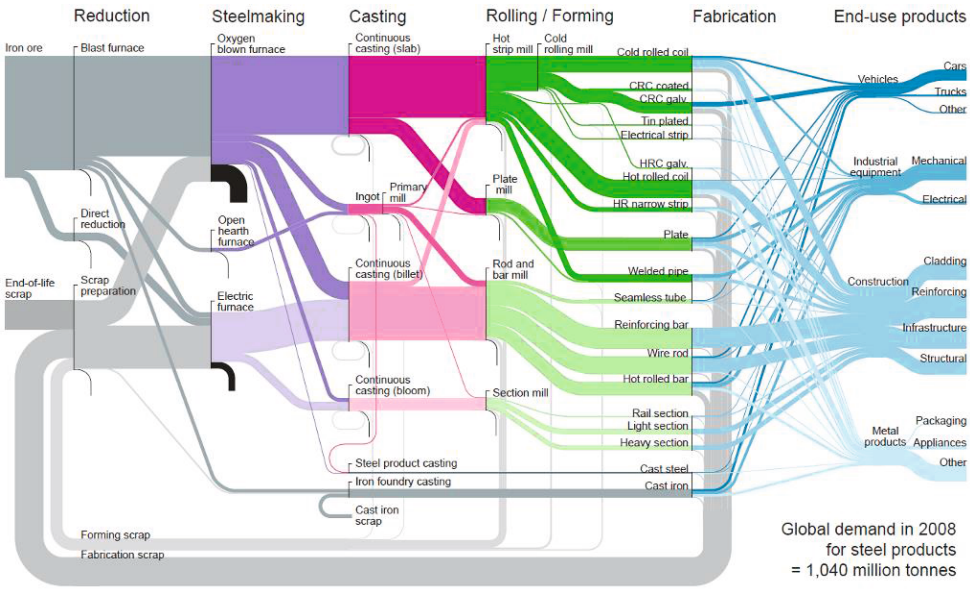


Figure 14 Global flow of steel from liquid metal to end-use good (Cullen et al., 2012).

Quality concerns for secondary production are mainly relevant for sectors that depend on high-quality steel, such as vehicles (Pauliuk et al., 2013a, Nakamura et al., 2012). The mentioned quality challenge is mainly due to tramp elements, such as copper and tin, which accumulate in the recycled material (Pauliuk et al., 2013a, Ohno et al., 2014). Figure 14 shows that most of the steel scrap ends in the construction sector, whereas vehicles are mainly recipients of primary steel due to the low tolerance of tramp elements for steel components. In addition, end-of-life vehicles are one of the major sources of copper contamination for iron and steel scrap (Nakamura et al., 2012, Igarashi et al., 2007, Ohno et al., 2014).

Primary steel is one of the most widely used materials within the automotive industry (Ducker, 2011b). Since the first car generations, mild steels, or as they are more accurately referred to, low-carbon steels, were dominantly used due to favorable properties, such as strength, formability, cost and design flexibility. However, the first oil crisis in the early 1970s and the adaptation of fuel economy standards in North America made the industry begin to seriously look toward light-weighting options and the substitution of higher-strength steels to improve fuel economy (Horvath, 2010). Car models in 1975 contained 56% (mild) steel, 4% medium- and high-strength steels (HSLA), 2% other steels and 15% cast iron. In 2007, mild steel contributed 43%, medium- and high-strength steel together with advanced high-strength steel (AHSS) 12%, other steel 2% and cast iron 7% (Ducker, 2011b). The same study predicts that North American light vehicles in 2015 will contain 34% mild steel, 10% AHSS and 8% HSLA, 2% other steel and 6% cast iron (Schultz, 2009, Ducker, 2011b). Figure 15

shows North American light vehicle iron and steel content in kg per vehicle for snapshots of 1975, 2007, and 2015.

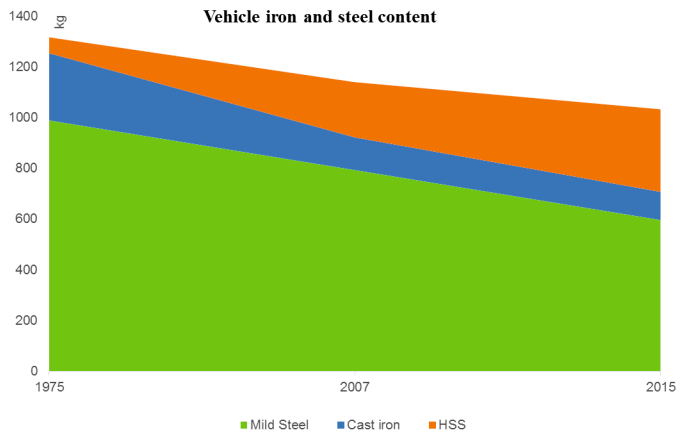


Figure 15 North American trends of light vehicle iron and steel content from 1975 to 2015 according to (Schultz, 2009).

1.4 Trends in ELV management

1.4.1 Current practices

Waste management is an important topic in environmental issues and, among all, the vehicle sector generates approximately 5% of the world's industrial waste (Simic, 2013). The recent decade's substantial growth in car ownership globally will lead to significant growth in the number of deregistered cars and ELV flows. Although the management of ELVs is required in every country, it is notably important that large countries and regions with high growth rates in their markets, such as China and India, are able to address future ELV challenges and complexities (Sakai et al., 2014).

The handling of ELVs is similar in most countries regardless of the legislative management system (Sakai et al., 2014). The process of ELV recycling generally starts with i) depollution, where hazardous substances, such as lead batteries, mechanical oils and refrigerant gases, are collected; ii) recovery of recyclables and materials suitable for secondary use, including dismantling of engines, tires, and bumpers; iii) remaining car hulks are shredded; iv) the shredded materials are sent to an air classifier, where the light automotive shredding residue (ASR), or so-called fluff, is removed from the remaining fraction; v) ferrous material, mainly metals, are removed by magnetic and eddy current separation (Sakai et al., 2014, Zorpas and Inglezakis, 2012, Gaustad et al., 2012); and vi) sink float or heavy media separation is used to separate non-ferrous materials with different densities; typically Mg, Cu, Zn, and Pb, can be sorted out from

aluminum in this step (Gaustad et al., 2012). Color sorting appears to be an effective way to sort shredded ELVs (Gaustad et al., 2012). Zinc, copper, brass, and stainless steel can be separated from aluminum in a non-ferrous scrap stream. Hand sorting is a prevalent practice in countries with low labor costs, such as India and China. It is estimated that hand sorting of aluminum automotive shred can achieve 99% accuracy (Gaustad et al., 2012). In addition, wrought and cast aluminum fractions can be sorted by hand due to distinctive surface characteristics (Gaustad et al., 2012, Rao, 2006). Color sorting can also be achieved by automated processes by analyzing images of each scrap piece and directing the pieces to different feeds. To further separate non-ferrous metallic fractions, chemical etching is used in combination with color sorting. However, this method requires using chemicals that may have additional environmental impacts; furthermore, the automated process is not yet cost efficient (Gaustad et al., 2012). There are other spectroscopy techniques that are used for the identification and sorting of shredded scrap; in particular, laser induced breakdown spectroscopy (LIBS) has shown great promise for sorting wrought and cast aluminum (Gesing, 2004, Cui and Roven, 2010, Gesing, 2006). However, this technology is still too expensive to compete in the market and also requires that the scrap be free of lubricant, paint, other coatings, and oxide formation.

1.4.2 Legislation

ELV management is becoming more important due to the increasing number of vehicles reaching the end-of-life and the increasing complexity of materials. A recent comparative study of end-of-life vehicle recycling system declared that legislative ELV recycling systems are established in the EU, Japan, Korea, and China, whereas in the US, ELV recycling is managed under existing laws on environmental protection (Sakai et al., 2014).

In the EU, the EU-directive 2000/53/EC on ELVs was enacted in 2000. The initiative for the EU-directive started in 1989, the year in which the European Commission set up a program of actions on 'priority waste streams,' including ELVs (Smink, 2007). In 1991, a European ELV-project group was established with representatives from different stakeholders that were identified in the process. The effort resulted in a set of key documents, such as a legislative proposal that focused on the synchronization of ELV legislation, among different national schemes for addressing ELVs. Later, the European Parliament called on the European Commission to legislate on waste streams, in particular ELVs, based on producer responsibility. These efforts lead to a proposal for a Directive on ELVs (COM (97) 358), which later resulted in the EU-Directive 2000/53/EC (Smink, 2007, European-Commission, 2000). The directive will ensure that all Member States have uniform legislation on the re-use and recycling of cars at the end of their useful life. The EU-Directive 2000/53/EC (European-Commission, 2000) set a target for the reuse and recycling rate of ELVs in the Member States, and states: i)

by 01/01/2006, the reuse and recovery rate should reach 85% on a mass basis (recycling 80%) for vehicles produced after 1980 and ii) by 01/01/2015, the reuse and recovery rate should reach 95% on a mass basis (recycling 85%). According to Zorpas and Inglezakis (2012), currently approximately 75% of ELV total weight is recycled, whereas the remaining 25% is ASRs. Therefore, the automotive industry may face a challenge in meeting EU 2015 environmental standards.

In Japan, the law for the recycling of ELVs was enforced in 2005. The act emphasized the specific components to be recycled, and the target is to recycle 80% of airbags and 85% of ASRs by 2015. Recycling fees are paid by buyers at the time of purchase, and these fees are deposited into the deposit management system (Sakai et al., 2014).

In Korea, the act for 'Resource Recycling of Electrical and Electronic Equipment and Vehicles' was enforced in 2008. Under the current act, the responsibility for ELV recycling is placed on all stakeholders, including manufacturers, importers, dismantlers, shredders, ASR recyclers, and refrigerant gas processors. The material recycling and energy recovery target is set at a minimum of 85% by 2014, including energy recovery of less than 5%, and at least 95% after 2015, including energy recovery of less than 10% (Sakai et al., 2014).

In China, recycling rates are to reach approximately 85% (or at least 80%) material recycling by 2010, approximately 90% (or at least 80%) by 2012 and approximately 95% (or at least 85%) by 2017 (Sakai et al., 2014).

In the US, ELV recycling has been promoted by the Automotive Recyclers Association. The rate of material recycling was reported to reach 80% (Kumar and Sutherland, 2009, Sakai et al., 2014).

In most European countries, 80% of the ELV total weight is recycled, and the remaining 20%, which is automotive shredder residue (ASR), is currently disposed in landfills. Therefore, reaching the required target of the EU directive requires a considerable increase in ASR recycling, which can be challenging using current practices (Zorpas and Inglezakis, 2012). Reuter et al. stated that by minimum dismantling, and advanced post-shredding technologies, recycling quotas of approximately 85% are difficult to achieve (Reuter et al., 2006).

All of the reviewed ELV management legislation around the world failed to consider quality aspects. They only set targets for recycling quotas but, for instance, alloy separation is not considered, which can have negative effects on the recycling of those metals with limited refining options, such as aluminum (see chapter *1.5.2*). Even a small amount of unrecyclable ELV scrap could pose a challenge to the fulfillment of the EU-wide ELV Directive.

It is likely that the degree of scrap contamination will increase in the future due to the increased use of more complex appliances for safety and comfort purposes (Igarashi et al., 2007) or due to the fate of material quality over time and across products in open-loop recycling systems (Nakamura et al., 2014, Nakamura et al., 2012).

1.5 Challenges for aluminum recycling

1.5.1 Current challenges: insufficient scrap

Models of the global aluminum cycle show that aluminum recycling is still highly constrained by the amount of post-consumer scrap (GARC, 2011, Rombach et al., 2012, Liu et al., 2013a). In 2010, approximately 48 MMt of aluminum entered the use phase while 11 MMt of post-consumer scrap was collected for recycling, whereas approximately 30 MMt of aluminum products were added to the global aluminum stock in 2010 (GARC, 2011, Liu et al., 2013b). Excluding packaging applications, aluminum is used mainly in sectors with long lifetimes, such as buildings, transportation and engineering. Accumulated primary aluminum production in the period from 1950-2010 was approximately 900 MMt, of which 700 MMt is still in the use phase (Rombach et al., 2012, Liu et al., 2013b).

Today, the utility of aluminum is maintained through blending of mixed scrap with other types of scrap and primary metal. As noted above, there are no signs of a flattening of aluminum stocks; however, once the fast stock growth declines and more of the already produced aluminum reaches the end-of-life, there will be a large potential for recycling and reducing raw material use. Nevertheless, as the share of post-consumer scrap rises, aluminum recycling is likely to face new challenges related to alloying elements and impurities in the scrap.

1.5.2 Future challenge: use all scrap and maintain quality

The second challenge aside from the increasing number of ELVs and consequent complexities in end-of-life management is contamination by alloying elements and refining difficulties in aluminum recycling (Nakajima et al., 2010, Van Schaik et al., 2004). Figure 16 is borrowed from a study of thermodynamic analysis of contamination by alloying elements in some base metals, such as Fe, Cu, Zn & Pb, Pb, and Al, recycling (Nakajima et al., 2010). This radar chart is an extended work on the concept of 'metal wheel,' which was introduced by Verhoef et al. (2004) and demonstrates the importance of understanding metal linkages in natural resource processing. It is specifically relevant for recycling possibilities of metals based on the thermodynamic behavior of alloying elements in the metal, slag, and gas phases of the base metals. The smaller grey circles denote typical additive elements. In the case of aluminum, the figure indicates that Mg, Ca, and Ba can be removed by oxidization (transferred to slag)

and that Zn, Cd, and Hg can be removed by evaporation. The removal of the other 39 elements is extremely difficult, as they tend to remain in the metal phase. In comparison with other metals, the removal of alloying elements is far more difficult for aluminum than for iron, copper, zinc, and lead (Nakajima et al., 2009). The difficulty of designing material specifications in the refining and recycling process narrows possibilities to use aluminum in recycling.

Cu, Fe, Mn, Si, and Zn are the most common alloying elements of aluminum (Nakajima et al., 2010). Casting alloys generally contain more alloying elements than wrought alloys and can therefore be produced from mixed scrap (McMillan et al., 2012), although they may require blending with other types of scrap or primary aluminum to reach the required concentration of alloying elements. Conversely, most wrought alloys contain fewer alloying elements in lower concentrations and therefore have a low tolerance for accepting alloying elements and impurities from mixed scrap (McMillan et al., 2012).

Consequently, if the scrap mixture is not sorted before melting, recycling depends on growing casting stock to absorb scrap.

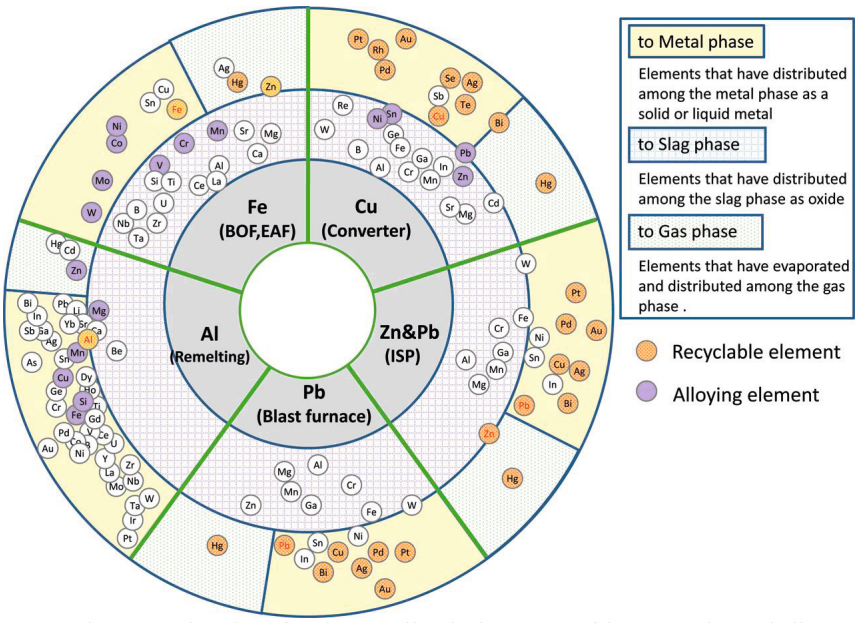


Figure 16 Element radar chart for the metallurgical process of base metals (Nakajima et al., 2010).

1.5.3 Role of vehicles in the global aluminum cycle

The study of aluminum recycling in the automobile industry has gained attention because passenger cars form a quality bottleneck in aluminum recycling. Passenger cars

embody most of the secondary castings, which are in turn the major recipients of recycled aluminum from all other sectors (Gesing, 2004, Furrer, 2010, Hatayama et al., 2007, Modaresi and Müller, 2012, Hatayama et al., 2012).

Aluminum use in passenger cars has been growing from 32 kg in 1978 to 149 kg in 2009 and approximately 156 kg in 2012 for average passenger cars (Ducker, 2009). Recent studies showed that one of the largest and fastest growing reservoirs of aluminum in-use resides in automobiles (IAI, 2006, EAA, 2008, Liu et al., 2013a). Today, the transportation market accounts for nearly 43% of the metal used in Japan and 35% of North American and West European aluminum shipments, whereas 40 years ago, transportation was responsible for only approximately 20% of the total consumption in the major car manufacturing countries of United States, Japan and Germany (Nappi, 2013).

Today, most of the aluminum is used in the powertrain, with 80-85% of cast parts that typically contain high alloying contents that can be produced from post-consumer or new scrap (European Aluminium Association, 2011, Furrer, 2010, Gesing, 2004).

It is generally expected that the largest growth potential for aluminum use in passenger cars is in wrought aluminum alloys and in components such as BIW, closures, bumpers and crash boxes, and suspension frames (Ducker, 2009, Ducker, 2011a, Ducker, 2012a, Hirsch, 2004, Gesing, 2004, Ducker, 2012b).

Figure 17 shows the common recycling paths (due to quantity and quality reasons) of aluminum scrap from different end-use sectors in a simplified manner. This current practice of cascadic use depends on the continuous and fast growth of the passenger car stock; otherwise, there will be a scrap surplus that cannot be absorbed by the automotive industry, if closed loop recycling into new vehicles is assumed. Consequently, the development of the global vehicle stock – in terms of size and composition – is crucial to the future demand of primary and secondary aluminum as well as the future supply of scrap from retiring vehicles. It is crucial to know the timing and amount of the future scrap surplus.

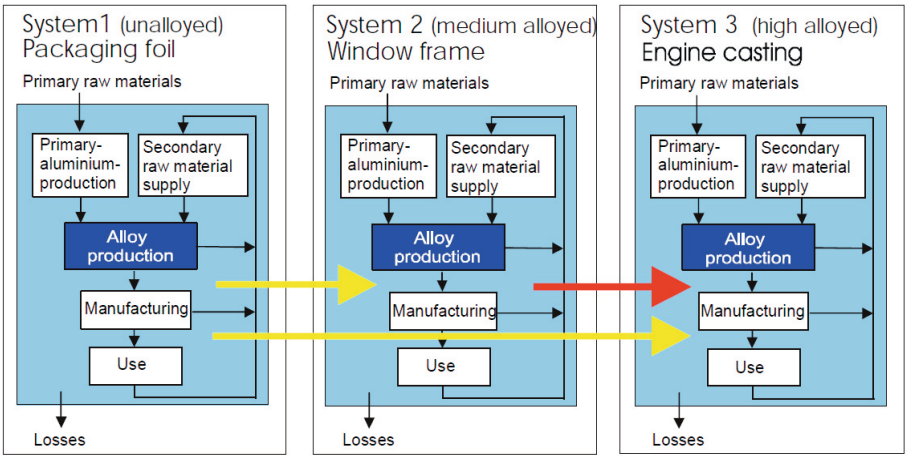


Figure 17 Interaction in open-loop aluminum recycling systems (Rombach, 2001).

1.6 Motivation for the development of global stock dynamics models

The future scrap surplus problem was already discussed over a decade ago. Zapp et al., (2002) analyzed the long-term supply of aluminum to the European automotive industry and compared future casting demand with scrap amounts from different sources, using historic production data and assumptions of future car market developments and use of aluminum in cars. This study estimated casting demand and scrap supply independently using trend analysis, thereby ignoring their connection through the dynamics of the vehicle stock. They found that old scrap availability would eventually exceed casting demand in 2040. Gesing (2004) conducted a study at the global scale that simulated the mass balance in the vehicle system based on historic data on global aluminum consumption in vehicles and global aluminum recycling from vehicles and other sectors. The study, which was only based on historic trends without any scenario development, concluded that there would be a scrap surplus that could not be absorbed by cast alloys production, which would be unsuitable for wrought alloys due to the high alloying elements concentration. However, the study was not able to predict the timing and extent of the future scrap surplus. These two studies were only based on studies of flows and therefore are unable to study the stock development and to analyze vehicle and aluminum stock dynamics. Therefore, such models are unable to calculate the expected scrap from end-of-life.

In-use stocks provide services to society. For instance, vehicle stocks provide mobility. Consequently, material stocks in vehicles provide services to society throughout the vehicle lifetime. A traditional view on materials is based on a production-driven approach (van der Voet et al., 2002, Brattebø et al., 2009, Baccini and Brunner, 1991,

Krausmann et al., 2009), in which a stock is calculated as a function of production and lifetime, but this method is insufficient to predict the future stock. In contrast, the stock-driven approach includes parameters (population, service per capita, and lifetime) that determine the stocks, whereas the flows are derived from the stock development (Müller, 2006, Bergsdal et al., 2007, Pauliuk et al., 2013b). A stock-driven approach is proven to be more robust compared to the flow-based intensity of use models because stocks reflect the ultimate demand for services in the built environment (Müller, 2006, Gordon et al., 2006, Müller et al., 2010, Pauliuk et al., 2011, Hatayama et al., 2009).

A dynamic MFA approach to model vehicle in-use stock and related aluminum recycling system has been used in some previous studies (van Schaik et al., 2002, Cheah et al., 2009, Hatayama et al., 2009, Hatayama et al., 2012).

Van Schaik et al., 2002 developed a dynamic model for passenger cars that defined the link between end-of-life vehicles and the recovery of various metals, including aluminum. They predict the number of ELVs in the Netherlands using different distribution functions for lifetime assumptions based on passenger car production data and combined the model with an optimization model for recycling, which calculates the recovery rate of aluminum as a function of variables with regard to different material streams, such as aluminum alloys found in scrap fractions. Cheah et al. (2009) employed a dynamic MFA model to calculate the annual stock and flows of aluminum in the US passenger cars from 1975 to 2035. The US vehicle stock is calculated based on historic vehicle sales and scrappage rates and future forecasts of the US market. The main intention was to analyze the corresponding energy embodied in automotive aluminum and cumulative aluminum production energy demand. These two studies calculate vehicle stock and were thus able to follow stock development and connect supply and demand. However, the main driver for the vehicle stock was production data; therefore, they were unable to predict the future based on demand from society.

Hatayama et al. (2009) estimated the amount and quality of aluminum in-use stock and scrap generation in different sectors, such as automotive, construction, beverage cans, and machinery, in Japan, China, the US, and Europe using a dynamic MFA. The concentration of alloying elements in stocks and flows were calculated by counting the consumption in each end use by alloy type. Then, from the relation between the amount of in-use stock and per capita GDP in the past, future in-use stock change was predicted using GDP and population forecasts. Later, in another study, Hatayama et al. 2012 employed a stock-driven dynamic MFA approach to the automotive sector for the same selected regions. The model was used to analyze how the recycling of aluminum will change by 2050 by introducing next-generation vehicles and scrap sorting. The model distinguished between wrought and casting aluminum. A comparison of demand with discard was used to evaluate the amounts of primary aluminum required and scrap that cannot be recycled because of a high concentration of alloying elements. The result of

their study showed that there would be 6.1 Mt of unrecyclable scrap in 2030 in the selected regions. This study still lacks a global scale, which is necessary to anticipate the timing of scrap surplus.

The reviewed previous studies demonstrate that there have not been any global stock-driven approaches that analyze global aluminum recycling challenges. Previous studies were either only trend analyses based on historic data that was extrapolated or only regional studies. Trend analysis is not sufficient to predict future material cycles because it is only based on historical production data. On the other hand, future stock changes are based on societal demands, which can only be captured by stock-driven models and by including demand parameters.

It is important to study the scrap surplus using a global scope because countries are open systems with trading of all relevant aluminum products along the cycle. If one region or country faces a surplus of scrap, it may export scrap to other regions with a scrap deficit; as a consequence, the problem will manifest on a global scale, provided that transport costs do not inhibit trade. If transport costs limit trade, scrap shortages would occur at different times in different regions; however, prices for sorted aluminum scrap have historically been close to primary aluminum prices, and transport costs have not been an important trade barrier.

In this thesis, a stock-driven approach is used to study the selected material cycle of the global passenger car fleet. Employing a stock dynamics approach for scrap surplus calculations allows for mass balance consistent estimations of aluminum demand, scrap availability, and stock in service.

In addition to the lack of information about the timing and amount of the future scrap surplus on a global scale, there was a lack of understanding regarding the most effective combinations of interventions to avoid a future scrap surplus. Some previous studies, such as Gaustad (2011), employed optimization or allocation models in addition to a dynamic MFA model to analyze options to mitigate the negative impacts of accumulation on scrap utilization. This study was able to allocate scrap and primary material to individual products. However, it is only possible to capture all of the required information by considering component and alloys levels within a product category. Therefore, there is a need for a detailed study that uses optimization models based on refined models.

To evaluate different solution strategies, it is necessary to simultaneously forecast scrap supply and aluminum demand on a component and alloy basis. An understanding of component levels and the alloying element resolution is necessary to quantify the capacity for scrap use. This information allows us to test whether the separated scrap

fractions could be used in components that contain alloys other than secondary castings aluminum.

The global emission reduction potential of light-weighting passenger cars depends on fleet development. Whereas light-weighting cars reduce emissions in the use phase, upfront emissions in the production phase may be intensified depending on the materials used and the share of secondary production (recycling), which may in turn depend on scrap from retiring vehicles.

Traditionally, LCA has been used to address questions related to emissions, material use and end-of-life management in vehicle systems (Kim et al., 2010b, Keoleian and Sullivan, 2012, Geyer, 2008, Mayyas et al., 2012, Das, 2005, Das, 2014, Stodolsky et al., 1995, Bertram et al., 2009a). A review of 43 LCA studies on the emissions benefits of light-weighting in automobiles were compiled in a publication from Kim and Wallington (2013). Typical LCAs consider functional units or single vehicles only, and practitioners must make assumptions on how the background economy evolves over the product's life cycle. This includes assumptions about material recycling within and outside the vehicle system, which in many cases is the single most important measure to reduce emissions from material production systems. To assess the possible overall emissions reduction, material challenges and potentials related to passenger car systems over the next decades, it is not sufficient to simply scale up the LCAs of single vehicles for the following reasons: (i) the vehicle stock is composed of different cohorts, which limits the rate at which new cars can penetrate the fleet; (ii) technological changes in vehicles and the material and fuel supplying industries must be considered; and (iii) changing material composition will change the scrap flows from end-of-life vehicles, which can impact the recycled content of new cars in the future and hence reduce embodied emissions. Furthermore, the dynamics of the system dictate the availability of scrap, and hence the possibilities for recycling, through the extended lifetime of vehicles. On a global scale, the level of recycling and the resulting emission savings are limited by the total scrap availability, which changes substantially over time. Some of these limitations are related to scale, and some are related to dynamics that can be overcome by modeling the entire vehicle fleet at a global scale with a dynamic MFA approach, including demand drivers (population and vehicles per capita), technological change, and material recycling. These aggregate dynamic effects have not been studied before. Therefore, there is a need for dynamic fleet-recycling models that allow us to assess specific technologies in a global setting. Such a model connects population estimates, lifestyle choices, and utilization parameters to inventories of specific drive technologies and material production processes. The scrap availability and sales potential for end-of-life vehicle scrap are taken for granted in LCA, whereas the dynamic fleet-recycling model showed that scrap availability changes over time, depending on several factors that can be made explicit.

There is a need for a model that is able to capture different cohorts in vehicle stocks while considering technological and material composition changes over time. Therefore, we developed a dynamic stock model of the global car fleet and combined it with a dynamic MFA of the associated steel, aluminum, and energy supply industries. This dynamic fleet-recycling model provides estimates of the emission saving potentials under different scenarios for the substitution of conventional steel with high-strength steel (HSS) and aluminum at different rates combined with recycling scenarios over the 2010-2050 period.

1.7 Goals and research questions

This thesis aims to fill the mentioned gaps of understanding the linkage between the global vehicle system and material cycles, such as the aluminum cycle, on a global scale over time. The main questions to be addressed in this PhD project are:

1) How are the dynamics of the global vehicle stock changing the boundary conditions for aluminum recycling?
2) What are the most effective interventions to minimize a future aluminum scrap surplus?
3) What material substitution options for vehicles exist to reduce direct and indirect GHG emissions over time?

2 Discussion and outlook

This section aims to answer the three research questions, and it discusses the strengths and weakness of the approaches and the implications of this thesis for policy and research.

2.1 Main findings and reflections on the research questions

2.1.1 Aluminum recycling and the dynamic of the global vehicle stock

The first proposed research question was:

- How are the dynamics of the global vehicle stock changing the boundary conditions for aluminum recycling?

The vehicle fleet is an important driver of aluminum demand and scrap generation. More importantly, the vehicle fleet is the key carrier for casting, accounting for more than 70% of the total casting demand. Therefore, it currently creates the main sink for aluminum scrap from all sectors. The global vehicle stock is growing substantially and this fast growth in vehicle fleets allows secondary casting in vehicles to absorb scrap. However, the study result shows that the scrap absorption capacity of secondary cast in the vehicle fleet may not grow sufficiently fast in the long term. This can be explained by 1) a decline in the growth rate of the vehicle fleet and 2) a tendency toward smaller amounts of secondary castings per vehicle due to changes in new powertrain technologies. Diesel and electric vehicles use approximately 20% and 50% less cast aluminum, respectively, compared to gasoline vehicles. Castings have a smaller potential for growth because their application in engine components has already penetrated the market to a high extent, and there is a trend to replace gasoline engines with new technologies, such as diesel, hybrid, and electric, which use less or no secondary casting. Starting from the current level of 100 kg, saturation levels of 80-130 kg are expected. 3) The growth potential for wrought aluminum in passenger cars is considerably higher than for castings, mainly due to promising new applications in body-in-white (BIW). Wrought aluminum growth is assumed to increase substantially from the current level of approximately 50 kg to 100-300 kg by the end of 2100.

Due to the different penetration rates of wrought and casting aluminum in vehicles, the rate of scrap generation is growing faster than the demand for secondary castings. The results confirm that the automotive aluminum sector may go from being a net scrap consumer to a net scrap producer in the coming decade.

Referring to the results of papers I and II, all scenarios reach a point at which the sum of the scrap supply from passenger cars and additional minimum aluminum resources for dilution exceed the secondary castings demand. The timing of the surplus occurs in

2018 for the base scenario and is relatively robust, from between 2012 and 2028, depending on different parameter variations. With the highest dilution rate assumption, the model shows the most extreme result, in which the surplus would occur in 2012. However, wrought separation in ELV management is the most effective parameter to delay the future scrap surplus until 2028.

The gap between demand and supply varies considerably under different parameter assumptions. The scrap surplus reaches a level of 3.3-18.3 million metric tons per year in 2050.

Higher population and car ownership can delay the time of the scrap surplus by one or two years; however, higher population and car ownership will create a significantly larger scrap surplus in the long term. An extension of vehicle lifetimes delays the scrap surplus by two years and keeps the scrap surplus at a slightly lower level.

According to paper IV's simulation, scrap surplus occurred from 2025, which is 7 years later than in the base scenario of our previous model. The main reasons for this variation are: 1) magnesium removal is considered for the base scenario in paper IV, whereas no refining option is considered in the model for papers I and II and 2) recycled content assumptions were fixed for papers I and II (56% for cast alloys and 0% for wrought alloys). However, in paper IV, the recycling content was changing and was determined by the chemical composition of scrap and alloys.

Regardless of this variation in the results of these simulations, the expected surplus is less than one vehicle generation in the future. Thus, the problem is largely determined by the vehicles stock currently in use and the consequent scrap generation rate, which reduce the uncertainty of the results. The new powertrain technologies affect scrap generation only in the longer term and thus have a minor impact on surplus timing; however, they may play a significant role in the magnitude of the surplus problem in the longer term.

2.1.2 Effective interventions and factors to overcome the scrap surplus

The second proposed research question was:

- What are the most effective interventions to minimize a future aluminum scrap surplus?

The model simulation showed that all sectors involved in the vehicle-related aluminum cycle could make contributions to solve the scrap surplus challenge. Figure 18 shows the most relevant interventions along the cycle. Aluminum recycling is becoming increasingly complex and requires multi-stakeholder governance. Although the

motivation for such interventions comes mainly from the aluminum industry, their realization depends mainly on stakeholders in other parts of the system.

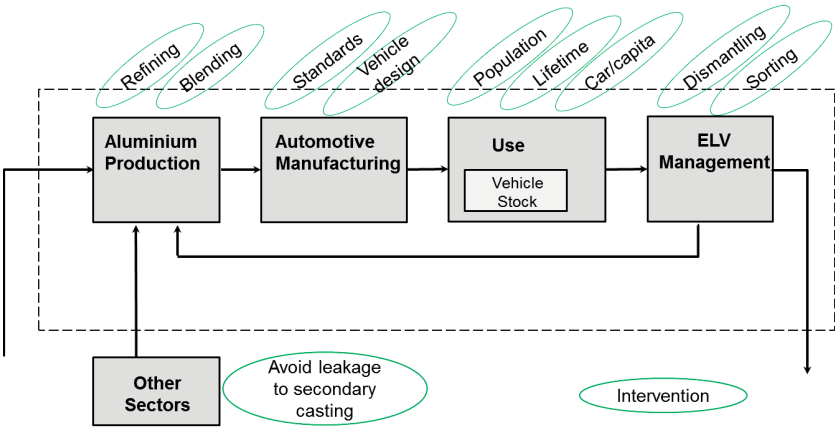


Figure 18 Relevant interventions to minimize a future aluminum scrap surplus.

The aluminum industry has two options to mitigate the scrap surplus problem directly; all other interventions require cooperation with other sectors. **Refining technologies** can be used for separating alloying elements and impurities from the aluminum melt (such as chlorination, electrolysis, fractional crystallization, hot crush, filtration, or floatation). They are currently expensive and have significant environmental drawbacks, including a high energy demand. For example, magnesium could be removed from molten aluminum scrap to achieve the low levels required in the most common secondary cast alloys. Due to the high value of magnesium, refining costs, and chlorine emissions, it is not the most desirable option. Reducing the dilution rate through the intelligent **blending** of different scrap alloys, is a cost-effective option to delay the scrap surplus. The effectiveness of the two latter interventions is limited, and as mentioned above. **Avoiding leakages to secondary casting** can be achieved by scrap recovery and sorting in nonautomotive sectors**,** such as packaging and building. This could reduce the amount of cascaded scrap currently being absorbed by automotive secondary castings.

In addition to the above options related to aluminum production, the two most effective interventions are related to the ELV management (increased dismantling and sorting), which can improve the quality of treated scrap and consequently widen the possibilities to recycle the scrap.

Increased sorting (automatic or manual) of mixed aluminum scrap into casting, wrought aluminum, and different alloy families has a high potential for avoiding excess scrap in the medium and long term. Additional alloy sorting of mixed shredded scrap requires further advanced technology development, such as laser-induced breakdown

spectroscopy (LIBS), with relatively fast and high market penetration. According to the model simulations, an excess of scrap can be avoided if the enhanced sorting technologies are deployed rapidly on a global scale. Our calculations show that rapid market penetration of these technologies is even more important than the starting time of their deployment. If the technology is introduced in 2012, but over a 20-year time horizon, the surplus time is not delayed significantly. On the other hand, if the technology starts to penetrate the market in 2017, but with a shorter technology uptake period of 10 years, the surplus can be delayed until 2028.

Increased dismantling of components could reduce the mixed scrap by approximately one third. Potentially attractive candidates for cost-effective dismantling include wheels, closures, suspension frames, heat exchangers, bumpers, and crash boxes. A strategy of increased dismantling is best confined with a strategy of component-to-component recycling, which also requires interventions in automotive manufacturing and aluminum production. Increased dismantling does not require new technologies and could be implemented rapidly, provided that dismantlers have appropriate incentives to do so. However, the use of these components for alloy-specific recycling is currently limited by the complex composition of components (mixed material design and applied joining techniques, i.e., steel bolts or rivets, different welding wire composition) and strict safety requirements from the automotive industry that practically prevent producing safety-relevant parts from recycled material. In some cases, such as wheels, component-to-component recycling may require new standards to enable their production from scrap. Currently, automotive manufacturers mandate strict recipes for the composition of aluminum alloys for safety components, which require the use of only primary sources. Instead, mandating properties would provide aluminum producers more flexibility in developing new alloy types that meet the required qualities with the use of obsolete scrap. This requires changing the **standards** in automotive manufacturing, and may therefore delay the implementation of a component-to-component recycling strategy.

In addition to cast alloys, there are few other alloys that can potentially absorb mixed scrap, such as alloy 6082, which can be a sink for a mixture of wrought alloys. Developing alternative sink alloys or recycling-friendly alloys in **vehicle designs**, which function as 'intermediate reservoirs,' can be an effective intervention for the future scrap surplus. Although intermediate reservoirs may not be the final solution to the alloy problem, they could be important in a transition phase by delaying the problem, because they can absorb some of the scrap and create a larger bottom reservoir for aluminum recycling while more advanced separation techniques are developed. Components can be designed to be more suitable for disassembly and recycling. This can be achieved, for example, by reducing the number of alloys employed in car components and reducing the material complexity in connections and joints. However, the desired effect would be delayed by approximately a vehicle lifetime, which is

insufficient to address the scrap surplus problem without additional measures in ELV management.

Changes in the use phase parameters also affect the timing of the scrap surplus. Higher **population** and **cars per capita** and longer vehicle **lifetimes** result in a several year delay of the scrap surplus. Interventions related to the use phase, population and cars per capita are not considered useful for relevant policy suggestions in this context because they may contradict several other environmentally friendly policies. However, extending the lifetime of a vehicle may be a relevant intervention that can be suggested to users.

2.1.3 Material selection in vehicles and consequences for global material cycles and energy use

The third proposed research question was:

- What material substitution options for vehicles exist to reduce direct and indirect GHG emissions over time?

This study showed that light weighting of passenger cars by material substitution could save between 9 to 18 gigatons of $CO_{2\text{-eq}}$ between 2010 and 2050. However, this range is considered an upper limit and its realization requires: (1) a rapid penetration of light-weight materials to their technically feasible potential by 2030 and (2) the utilization of the secondary mass saving potential in which lighter and smaller powertrains and other components can be produced as a result of primary mass saving while still keeping the same functionality. In addition, a potential increase in the mass of other vehicle components due to higher safety standards or more luxurious features (counter-effects), is considered.

Approximately 85-90% of GHG emissions occur during the use phase of vehicles (direct and indirect emissions) and 10% comes from the material production phase. Despite the lower emissions from material production compared to the use phase, the study of material substitution in global vehicle fleets is of high importance due to the light weighting benefits they provide and consequently the reductions in the use phase emissions. In addition, the relation between material supply, material efficiency, and recycling has long-term effects on GHG emissions.

The full benefits of currently available light-weighting options can be realized only in a few decades, when the global vehicle fleet is replaced by lightweight vehicles. In addition, the full benefits of recycling will become apparent only once the currently rapidly growing in-use stock of aluminum flattens and the scrap from ELVs becomes available for recycling. The effect of recycling on emissions is more important for aluminum than for steel because the relative savings are higher for aluminum and

because the aluminum stock in vehicles is growing faster than that of steel. Looking at the near future, up until 2025, the material choice for light-weighting is less important because savings in the use phase are higher than the material production emissions, but in the long term, aluminum has a higher potential to reduce emissions than HSS, providing that there will be significant improvements in the aluminum recycling system.

For the allocation of emissions to different metals, it is important to consider the exchange of scrap between the sectors. In the case of aluminum, the vehicle stock is a net sink of scrap from other sectors (mainly construction). In contrast, vehicles are a net source of steel scrap, which is used mainly to produce steel for the construction sector, which has lower quality requirements. Primary metal production and internal recycling can be expected to dominate the supply of automotive steel due to high quality standards.

In our model, which is limited to the automotive sector and related material sectors, we solved this allocation problem by assuming that emission benefits are only allocated inside the system boundary, and as long as the scrap is used for vehicle production, emissions benefits are assigned to the vehicle system regardless of which sector was the source of this scrap. In the same manner, where steel scrap left the system boundary for the construction sector, the vehicle system would need primary materials instead and therefore lost the benefits of recycling. Allocating emissions benefits of using secondary material is only possible with multi-sectorial analysis by considering all of the material transitions between sectors.

2.2 Methodology discussion

The results of papers I and II indicated that new recycling strategies would be needed to avoid a future scrap surplus. The study was the first scrap excess estimation that was an explicit system definition with transparent model assumptions. The dynamic MFA model used in papers I and II allowed for a robust identification of the scrap surplus problem. However, the crude resolution of alloys (casting versus wrought) and the lack of a component resolution result in high uncertainties and do not allow for a practical evaluation of interventions, such as the dismantling of components. The model cannot simultaneously forecast scrap supply and aluminum demand both on a component and alloy basis, which is necessary to test whether the separated scrap fractions could be used in components that contain alloys other than secondary castings aluminum.

Therefore, in papers III and IV, the previous dynamic MFA model used detailed additional data distinguishing 5 car segments, 14 car components, and 7 alloy groups to track aluminum alloys in component groups. In addition to the development of a detailed model, a source-sink diagram was developed to identify potential alloys that could serve as alternative sinks for the growing scrap supply. The source-sink diagram

and model were used in combination with expert consultation to discuss alternative ELV management strategies on a component-by-component level from a chemical, quantitative, and practical perspective. The solution strategies are evaluated in papers III and IV.

Paper IV employs a model with alloy chemical element resolution, combined with an optimization procedure to quantify the scrap surplus and recycling paths under maximum scrap utilization. This paper focused mainly on intervention options in industries (ELV management, secondary aluminum industry, and component manufacturing).

In paper V, the model for aluminum in vehicles was extended to include steel as a second material, as well as energy use for material production and vehicle operation. The vehicle fleet was divided into ten drive technologies (conventional gasoline, gasoline hybrid, conventional diesel, diesel hybrid, plug-in hybrid gasoline, plug-in hybrid diesel, electric, natural gas, H_2 combustion, and H_2 fuel cells) and five different fuel types were considered (gasoline, diesel, electricity, natural gas, and hydrogen). Annual kilometrage and age-cohort-technology-specific fuel efficiency were used to determine the total fuel demand. The material layer includes a dynamic MFA of the key automotive elements: steel (divided into cast iron, standard steel, and high strength steel) and aluminum (cast and wrought aluminum). The following processes for secondary material production were considered: (1) recycling (cascading) scrap from other sectors for use in automobiles, which can typically only be used to produce aluminum castings; (2) scrap use within the automotive sector; and (3) automotive scrap that is exported to other sectors due to quality constraints, such as construction.

The dynamic fleet-recycling model that was developed in paper V allows us to assess specific technologies in a global setting. The model connects population estimates, lifestyle choices, and utilization parameters to inventories of specific drive technologies and material production processes. It can help to design portfolios of emission mitigation strategies that bridge the gap between product-specific strategies and global emission reduction targets or benchmarks. Energy and material supply, energy and material efficiency, and lifestyle changes can be included. The model includes material production (primary and secondary), car manufacturing, use (vehicle fleet), and end-of-life vehicle management. Material recycling can only be understood properly from a fleet perspective. The comparative success of a certain emissions mitigation strategy was determined by system-wide emergent effects, such as the potential for material recycling, and not by individual material choices or product designs.

The main limitation in paper V relates to the focus of the study, which was the vehicle sector, and the model only able to show how increased recycling from introducing different recycling scenarios could reduce emissions inside the system boundary. From a

direct emissions perspective, the automotive aluminum industry benefits from scrap generated by other sectors, whereas the automotive steel industry cannot even benefit from recycling its own scrap. In addition, the aluminum scrap surplus and consequently the limited recycling capacity were not considered in this paper because the effect was related to outside of the defined system boundary. The model was able to calculate this effect; however, assumptions were simplified. If it was considered in the model calculation, the impact would be a reduction in overall emissions benefits in cases in which scrap surplus would occur because the scrap would not be used in any other applications. The main concern in such studies is how the system boundary should be defined and the emission benefits should be allocated.

In general, the models' limitations for other papers are also related to the resolution of the system definition and data availability. The main limiting factor in this study is the focus on only one sector (vehicles) and the definition of the system boundary only for the vehicle system. Therefore, the model is unable to capture the exchange of material and study the effects of interactions between sectors in relation to material cycles and emissions allocations. To understand the full consequences of the border shift between sections, a model including all of the relevant sectors' interactions inside the system boundary with economic considerations would be required. However, extending the system boundary would require enormous amounts of bottom-up data that are extremely difficult to extract, especially in sectors that have large regional differences and practices. Gathering bottom-up data from vehicle systems as the most harmonized sector at a global scale was still a significant challenge. In some cases, data were gathered from meetings with several experts in the aluminum, steel and automotive industries. As such, data gathering for relevant aluminum components in passenger cars and the composition of different alloys in each component can be mentioned.

Although it was essential to study the aluminum recycling issue at the global scale to capture the overall picture and avoid trade complications, if the markets for aluminum-containing products (including scrap) do not function perfectly, we might expect that a scrap surplus could occur in some areas earlier than in others. Furthermore, the parameters used in the model have strongly differing uncertainties. Nevertheless, the parameter variation showed that the result regarding the timing of the scrap surplus is relatively robust and the robustness of the model results partly from the fact that the expected surplus is less than one vehicle generation in the future. Thus, the problem is largely determined by the vehicle stock currently in use, not by hypothetical future vehicle stocks, for which uncertainties are considerably larger.

2.3 Implications of the work for research and policy making

Legislation for ELV management around the world focuses on the total amount of scrap to be recovered from ELVs for specified target years. The legislation is limited because

it is exclusively focused on the total quantity of materials to be recovered. It is important to measure the success of recycling strategies according to existing regulations and to make improvements in the policies. Vehicle-fleet recycling models allow for an increased focus on quality aspects to ensure that the recovered scrap can be used. Through studies of the dynamics and patterns of stocks in use, the timing and extension of the probable oversupply of post-consumer scrap can be captured. In addition, the most effective interventions could be found. Furthermore, policymakers to set realistic targets given the linkage between ELV management and the coordination of interventions between the sectors needed to solve recycling challenges. The results in the global context provide guidance for the aluminum industry and decision makers on when and how new ELV management strategies would need to be adopted to make use of increasing amounts of post-consumer scrap.

Papers I-II showed that there will be challenges in meeting the recycling targets in ELV management in the future. Papers III and IV provided strategies for ELV management that can avoid or delay the scrap surplus of aluminum.

The effectiveness of emissions mitigation strategies, including material substitution, depends on the evolution of the recycling system. Current policies, such as the US and European regulations, focus on reducing the tailpipe emissions of new vehicles. Current LCA research suggests that emissions reduction policies should not focus on tailpipe but on lifecycle emissions (Kim and Wallington, 2013). These studies have highlighted the importance of using a lifecycle approach to avoid merely shifting the problem from direct emissions in the use phase to emissions in the production phase. Although this is an important aspect that is neglected in current policies, LCAs with a single car perspective have severe shortcomings; they cannot capture changes in the recycling system, which have substantial impacts on industrial emissions. Dynamic effects of the different material cycles, especially the recycling potential, must be anticipated and included in emissions reduction policies. Therefore, beyond the common LCA, an alternative approach that considers vehicle fleet development over time and the implications for material cycles is suggested. The dynamic fleet-recycling approach models the impact of current consumption on the future recycling potential and analyses the relative effect of different interventions along the entire system over time. This study can be informative for policy makers to anticipate future challenges in the end-of-life vehicle management.

Models, such as the one applied here, can help to design emissions mitigation strategies that connect product-specific strategies to sector- and economy-wide emissions targets. Focusing on one sector represents a significant limitation in emission allocations. Therefore, expanding this type of study to other relevant sectors and increasing the system boundary to include linkages between sectors are suggested.

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