Life Cycle Greenhouse Gas Emission Assessments of Automotive Materials

The Example of Mild Steel, Advanced High Strength Steel and Aluminium in Body in White Applications

Methodology Report

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1.1 Introduction

The environmental performance of materials can only be assessed within the context of their application. This study is thus concerned with greenhouse gas emissions (GHG) from automotive vehicles rather than automotive materials alone. In principle, this includes all emissions that contribute to climate change from any process necessary to produce, use and retire vehicles, i.e. all GHG emissions that are emitted during the so-called vehicle life cycle and attributed to the vehicle. This definition of automotive GHG emissions includes but goes beyond tailpipe CO_2 emissions of vehicles. There are many different drivers of GHG emissions from vehicles. For example, how vehicle owners use and maintain their cars has a large impact. However, the design of vehicles determines their GHG emission potential to a substantial degree, and it is thus imperative that GHG emission reduction objectives are already considered at the design stage. Automotive engineers and designers have a whole range of design options that might have the potential to significantly reduce automotive GHG emissions. Most of the options discussed in literature and the public debate focus on the use phase of the vehicle. Examples are reductions of aerodynamic drag and rolling resistance, engine modifications like variable valve timing (VVT) and variable displacement, charge modifications like direct fuel injection and turbochargers, fuel changes like liquefied petroleum gas (LPG), biofuels and hydrogen, transmission modifications regarding the number of gears and shifting schedules, and overall power train modifications like hybrid designs (CARB 2005).

The design strategy that is the subject of this report is vehicle mass reduction based on material substitution, specifically the use of aluminium and advanced high strength steel (AHSS) to replace mild steel. Obviously, there are other potential materials and design strategies to achieve vehicle mass reduction, like magnesium, fibre reinforced plastics or the use of smaller platforms or more efficient packaging. Mass reduction is also just one of many design strategies aimed at automotive GHG reductions. The focus of the study is thus rather narrow and highlights only one aspect of the broader agenda of GHG emission reductions from the automotive sector. This report does also not contain any technical and economic evaluations of vehicle mass reduction based on material substitution. Scientific validity and policy implications of this work thus have to be assessed with its intentionally narrow scope in mind.

The design focus on the use phase is unsurprising, seeing that the percentage of the life cycle GHG emissions of a vehicle emitted during its use and the associated fuel production and delivery typically ranges from 70% to 90% (see e.g. Sullivan & Cobas-Flores 2001, Schmidt et al. 2004, DBJ 2004 (Figure 1)). However, this does not mean that the other stages of the vehicle life cycle are irrelevant. Changes in vehicle design usually cause GHG emissions changes at several stages of the vehicle life cycle. These GHG emission changes may be of the same order of magnitude, regardless how disparate the absolute GHG emissions of those life cycle stages might be.

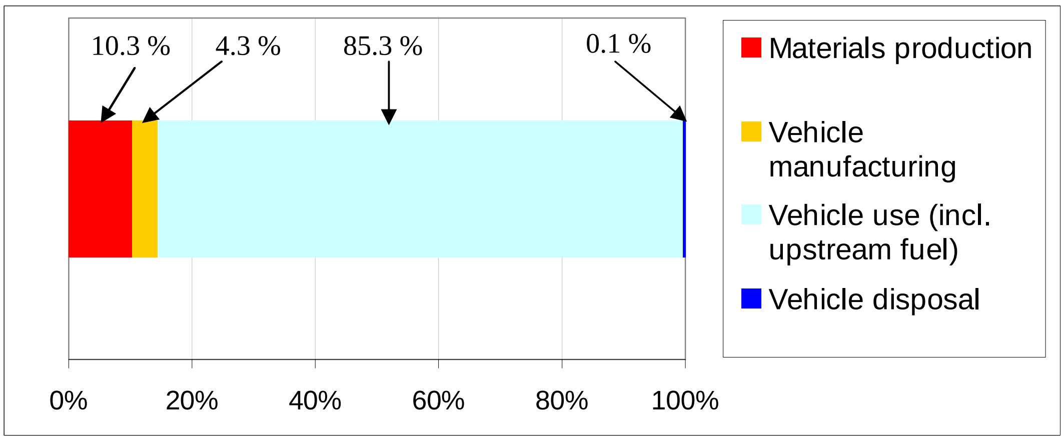

Figure 1: Typical life cycle GHG emissions of an ICE passenger car (Source: DBJ 2004)

An observation that corroborates the need for a life cycle perspective is that successful emission reductions of the use phase are likely to reduce its relative importance as can be seen in Figure 2. Here, the vehicle from Figure 1 is shown in its full hybrid electric version. The use phase share of the life cycle GHG emissions of the vehicle decreases, while the shares of all other life cycle stages increase. The same is true for emission reductions due to the use of fuels that have, relative to gasoline or fossil diesel, lower well-to-wheel GHG emissions per driven kilometre, such as certain types of ethanol and biodiesel (IEA 2004, Delucchi 2006). All measures that reduce GHG emissions from fuel production, delivery and combustion per driven kilometre, increase the relative importance of all other life cycle stages in the vehicle life cycle, which increases the importance of a life cycle perspective in the assessment methodology.

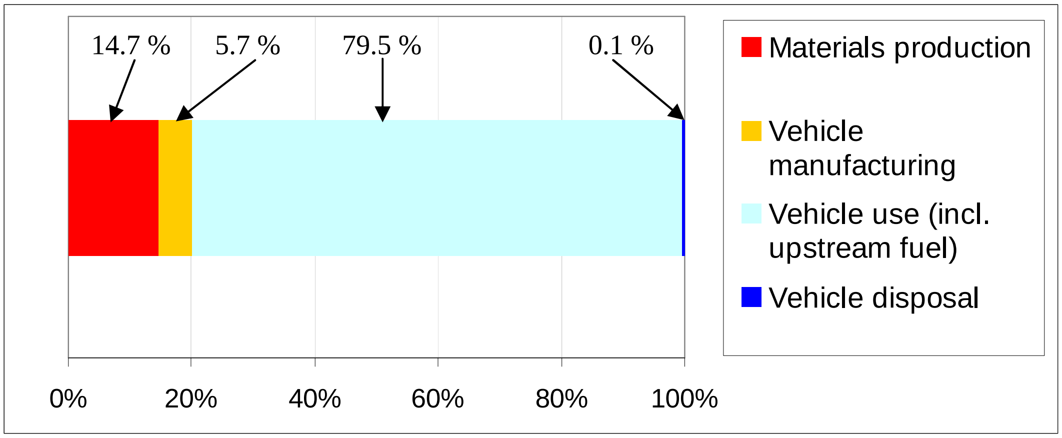

Figure 2: Typical life cycle emissions of a hybrid electric passenger car (Source: Own calculations based of DBJ 2004 and Bren School 2005)

1.2 Assessment Methodology

The assessment methodology presented in this report is based on life cycle assessment (LCA) according to ISO 14040/44 (2006). LCA is a technique compiling a quantitative inventory of relevant inputs and outputs of a product system; evaluating the potential environmental impacts associated with those inputs and outputs; and interpreting the results of the inventory and impact phases in relation to the goal and scope of the study (ISO 2006a). The presented methodology also emphasizes parametric modeling of the vehicle life cycle and therefore has similarities with previous studies such as Sullivan & Hu (1995).

According to ISO, LCAs investigate product systems with regard to a broad variety of environmental impact categories, such as resource depletion, ozone depletion, acidification, eutrophication, photochemical oxidation, toxicity and also climate change. This raises the possibility that changes in the product system decrease some environmental impacts but increase others; e. g. reducing ozone depletion but increasing climate change. Since there is no scientific method of reducing LCA results to a single overall score, such trade-offs across environmental impacts can not be assessed scientifically but need to be evaluated based on the relative societal importance of environmental impact categories, i.e. value judgments. Because the methodology advanced herein addresses only climate change, trade-off dilemmas do not arise. However, this methodology could be generalized to include multiple impact categories, in which case trade-offs would be possible. The product system, in our case a vehicle life cycle, is only assessed in terms of its potential impact on global warming, which is done through the well-established concept of radiative forcing (IPCC 2001). This scientific concept makes it possible to quantify and aggregate all relevant emissions, the so-called GHG emissions, in terms of their Global Warming Potential, measured in kg CO_2eq. Due to its narrow focus on climate change, the reported methodology is not an LCA according to ISO but an assessment of the impact of material choice on automotive life cycle GHG emissions based on LCA methodology.

Another central aspect of LCA methodology is its emphasis on the life cycle of a product. It is not just the use of vehicles that causes GHG emissions but all its life cycle stages, including material production, vehicle manufacturing and vehicle end-of-life (EOL) management (see Figure 3). Adopting a life cycle perspective is critical for the subject of this study since changes in the material composition of vehicles, such as using aluminium or AHSS instead of mild steel in body in white applications, may decrease the global warming potential of the use phase at the expense of increasing the global warming potential of the material production stage. It is precisely these potential trade-offs within one environmental impact category but across different stages of the vehicle life cycle that are the focus of the assessment methodology presented in this report.

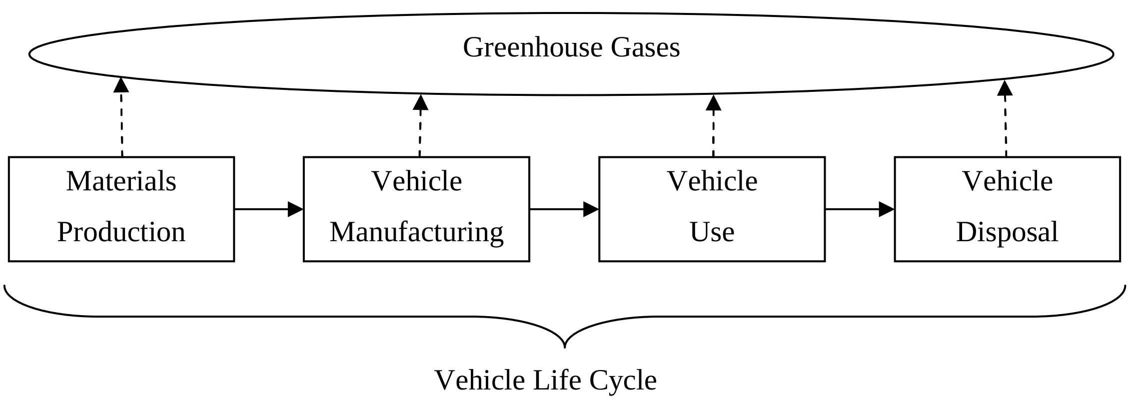

Figure 3: Assessing vehicle GHG emissions requires a life cycle perspective

A methodological distinction needs to be made between so-called attributional and consequential LCAs. Whereas attributional LCA assesses a product system in a given state, consequential LCA aims at quantifying the environmental impacts of a change in the product system (Ekvall & Weidema 2004, Ekvall 2006). This results in significant differences in the required methodology.

Attributional LCA requires allocation of elementary flows between different product systems whenever there are product flows crossing the system boundaries of the investigated product system. Clause 4.3.4.2 of ISO 14040/44 (2006) recommends avoiding allocation wherever possible through further division of unit processes or expansion of the product system. In order to meet its objective, i.e. quantifying the environmental impacts of a change in the product system, consequential LCA requires system boundaries that include all processes that experience significant changes due to the studied change in the product system. Changes in product flows crossing the system boundaries thus indicate that the boundaries have not been chosen correctly, since consequential LCA needs to account for any significant change in elementary flows caused by studied change in the product system, regardless where the elementary flows occur. This eliminates the need for allocation, but can come at the expense of far-reaching system boundaries.

Another important difference exists regarding the process data used to derive and quantify the elementary flows of each process in the product system. The data used in attributional LCAs typically describe the processes of the product system in a given state, say, how steel and aluminium is produced at the moment. Process inventories are therefore typically provided as averages of a certain technological, temporal and geographical coverage. Most process inventories are also modelled as linear functions of the economic output levels of the processes, i.e. doubling economic output doubles all elementary flows. Change-oriented consequential LCAs need to quantify the changes of elementary flows due to the changes in process technology and economic output levels that would follow from the investigated change in the product system. An example would be a world-wide increase in aluminium production due to a large-scale penetration of rolled and extruded aluminium in the automotive sector. For this reason, consequential LCA requires process inventory models that accurately reflect how changes in process technology and economic output levels change the related elementary flows.

The rigorous and systematic methodological distinction between attributional and consequential LCA has emerged only recently (Curran et al. 2001, ISO 2006a). It is fairly usual to find LCA studies that contain elements of both methodologies, such as attributional studies using consequential system expansion (see next two paragraphs for more discussion). Consequential life cycle inventory (LCI) modelling is significantly more complex than attributional modelling based on average process data and simple allocation or system expansion rules. Like all other assessments of the GHG impacts of automotive material choice known to the author, the presented methodology is based on average process data. Unlike most previous studies, it supports a considerable range of allocation / consequential system expansion choices. This approach is a natural first step towards more complex consequential models of unit processes, process relationships and system boundaries, the impact of which should be studied in future research.

In general, the need for allocation in attributional LCAs is caused by so-called co-production, i.e. when the product system or a unit process has two or more economic outputs (ISO 2006a). Examples are the distillation and cracking processes at petroleum refineries. The basic issue is the following: Assume that a unit process generates *m* kg of CO_2eq and has *x* kg of economic output A and *y* kg of economic output B. The studied product system only uses economic output A, while economic output B is used in other product systems. How much of the *m* kg of CO_2eq are attributable to economic output A and should thus be allocated to it? Allocation has been one of the major challenges in the development of LCA methodology, and the principles and procedures published in ISO 14041 (1998) (now contained in ISO 14044 (2006)) recommend that allocation should be avoided where possible through further division of unit processes or expansion of the system boundaries.

Automotive scrap recycling causes the following allocation issue: The vehicle itself is not the only economic output of the vehicle life cycle; it also generates prompt and end-of-life scrap as economic outputs, which typically leave the system boundaries of the vehicle life cycle. The question is now how to allocate the GHG emissions of the vehicle life cycle between the vehicle itself and the scrap outputs. The same is true for scrap inputs into the vehicle life cycle through the use of scrap-containing metal in vehicle production. Just like product flows in general, economic scrap flows across the chosen boundaries of the vehicle life cycle generate the need to allocate the inputs and outputs of process inventories or to further expand the boundaries of the investigated product system. Clause 4.3.4.3.1 of ISO 14044 (2006) states that the generic allocation procedures from Clause 4.3.4.2 also apply to reuse and recycling. It also states that changes in the inherent properties of materials shall be taken into account. It does not further specify what constitutes a change in the inherent properties of materials.

Clause 4.3.4.3.3 makes a distinction based on so-called closed-loop and open-loop recycling. According to Figure 2 in ISO 14044 (2006) the technical definition of closed-loop recycling is "Material from a product system is recycled in the same product system". 'Same' has to be interpreted as 'the same kind of', since material can not be recycled into the product it physically came from. ISO 14044 (2006) does not further specify what constitutes the same or a different kind of product system. The usefulness of a distinction between closed-loop and open-loop recycling is unclear since ISO 14044 (2006) also states that the distinction between closed- and open-loop recycling is irrelevant with regard to allocation. According to the standard, the only relevant issue is whether recycling changes the inherent properties of the material or not.

According to Clause 4.3.4.2 of ISO 14040/44 (2006), system expansion means "expanding the product system to include the additional functions of co-products […]". The standard does not distinguish between attributional and consequential methodology. However, there are two ways in which system expansion can be applied (Guinée 2002), which can be interpreted as attributional and consequential system expansion. System expansion in attributional LCA simply means the inclusion of additional processes in a given state, e.g. recycling of automotive scrap into metal products. As a result, the functional unit of the product system is also expanded (see e.g Saur et al. 1995). System expansion in consequential LCA means the inclusion of additional process changes, e.g. increased secondary metal production and decreased primary metal production due to increased scrap supply. The consequential type of system expansion is also called avoided burden method (Guinée 2002, Frischknecht 2006).

To be able to use the avoided burden method for product systems with recycling it is necessary to determine the exact effects of increased/decreased scrap generation or use. The challenges related to this have been pointed out in literature (Ekvall 1999, Ekvall & Finnveden 2001, Weidema 2001, Guinée 2002). One reason for these challenges is the fact that the relationship between increased scrap collection and decreased primary production is of socioeconomic rather than physical nature and thus based on socioeconomic rather than physical causality. In principle, an increase in scrap collection in a product system can lead to reduced primary production of the investigated material, reduced scrap collection elsewhere, an increase in the sum total of secondary and primary production of the investigated material, or reduced production of materials other than the investigated material (see e.g. Ekvall 1999, Ekvall & Finnveden 2001). Some LCA experts therefore caution against the indiscriminate use of the avoided burden approach, especially in attributional methodology (see e.g. Boustead 2001, Frischknecht 2006). Using the avoided burden approach in attributional life cycle inventory modelling also constitutes a mixing of methodologies, which may or may not be problematic.

Clause 4.3.4.1 of ISO 14044 (2006) states that the sensitivity of the results with regard to the chosen allocation method has to be explicitly assessed and communicated: "Whenever several alternative allocation procedures seem applicable, a sensitivity analysis shall be conducted to illustrate the consequences of the departure from the selected approach". As argued previously, the use of the avoided burden approach, i.e. consequential system expansion, has the risk of introducing significant uncertainties into attributional LCA. The methodology presented in this report is thus designed to facilitate sensitivity analysis of the effects of scrap inputs and outputs into and from the vehicle life cycle, regardless whether allocation or system expansion is used.

1.3 Automotive Materials and GHG Emissions of Vehicles

Material choices in vehicle design potentially impact GHG emissions at all stages of a vehicle life cycle (see Figure 4), the most obvious being the materials production stage. The global warming potential of economic material is typically given as so-called cradle-to-gate GHG emissions in kg CO_2eq per kg of material, which includes all upstream production processes (mining, smelting, refining, etc.). There are not only large variations in cradle-to-gate GHG emissions between different materials but also between different production routes, technologies and sites for the same type of material. It is important to make a conscious decision about how specific or generic the used process inventory data should be and to what extent these choices impact the results. The cradle-to-gate emissions of aluminium production, for example, vary widely between primary and secondary production routes and also strongly depend on the choice of production technology and inputs, like the type of anodes and electricity mix used in the Hall-Heroult process. Secondary production routes of metals typically have much lower GHG emissions than their respective primary production routes. The presented methodology thus explicitly discriminates between primary or secondary production routes of the used automotive materials.

For allocation or system expansion purposes, it also distinguishes between the GHG emissions of all material production processes up to slab/ingot level and those involved in further processing, like rolling, extruding, shape casting or galvanising. The material production stage includes all main processes to make finished or semi-fabricated materials (see Figure 5). Depending on the material product, the last process accounted for at the material production stage is rolling, galvanising, extruding or shape casting. Allocation due to the generation of co- or by-products during material production, including all metallic by-products (called home scrap), is typically already included in the available process inventory data (IISI 2002, IAI 2003).

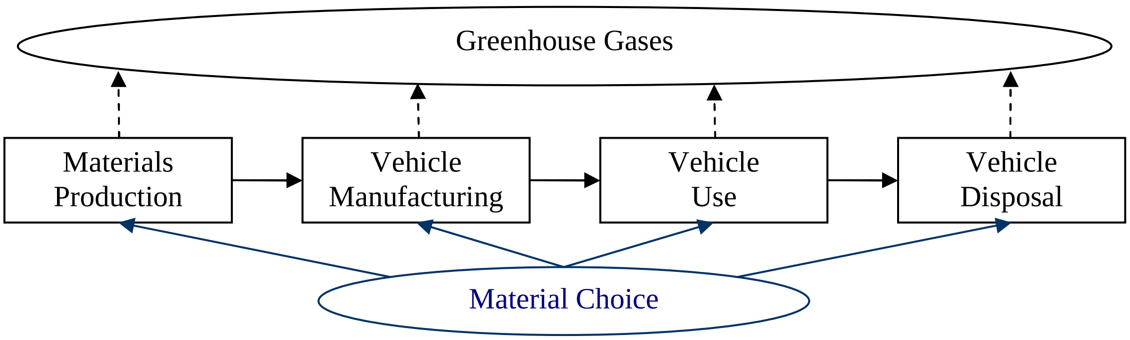

Figure 4: Material choice can impact GHG emissions of all vehicle life cycle stages

Due to mostly unavoidable inefficiencies in vehicle manufacturing more material needs to be produced than is contained in the vehicle. The amount of material required in the manufacturing process is called shipped material. The part of the shipped metal that does not end up in the vehicle is called prompt, pre-consumer or manufacturing scrap. Prompt scrap is typically easy to recycle due to ease of separation and low contamination. Prompt scrap generation can be specified either through prompt scrap rates or manufacturing yields, which are calculated as 1 minus prompt scrap rates. GHG emission allocation or system expansion due to prompt scrap generation at vehicle manufacturing is modelled explicitly in the presented methodology.

Different materials can also require different manufacturing processes, like forming, joining or coating technologies, which can also contribute to differences in GHG emissions. However, vehicle manufacturing contributes a relatively small percentage to the total GHG emissions of vehicles (Sullivan et al. 1998, Sullivan & Cobas-Flores 2001, DBJ 2004, Bren School 2005). The presented assessment methodology therefore does not account for the GHG emissions of material forming, joining or coating beyond the processes contained in the material production stage (see previous paragraph).

GHG emissions from the vehicle use phase are dominated by fuel production, delivery and combustion. The emissions of other aspects of the use phase, like vehicle maintenance and repair, are typically small (see e.g. Sullivan et al. 1998). This means that the single most important influence of material choice on use phase GHG emissions is its impact on vehicle fuel economy, even though it may also impact other aspects like ease of repair. A possible exception might be the use of automotive material that can not be repaired at all and has to be replaced after damage.

The only relationship between material choice and vehicle fuel economy that is studied in literature and known to the author is the mass savings potential of the material. Reducing the mass of a vehicle will improve its fuel economy, other things being equal. Even though the relationship between vehicle mass reduction and fuel economy improvement is very important, many studies use simple rules of thumb such as the so-called 5%-10% rule, which says that every 10% of saved vehicle mass improve fuel economy by 5% (An & Santani 2004).

Research carried out by FKA, a German automotive research institute, shows that the relationship is much more complex than this simple rule suggests (FKA 2005). The use of a constant mass elasticity of the fuel economy, like the 5%-10% rule, means that for vehicles of different baseline mass and fuel economy the same mass reduction results in different absolute fuel economy improvements. Assuming a constant elasticity, i.e. that the ratio of the two percentage changes is constant, also creates some fundamental methodological problems (see Section 2.4.2 for a more detailed treatment of this issue).

To avoid these problems, this study uses the more direct concept of absolute fuel savings (in l/100km per 100kg mass savings), which better reflects the fact that all resistance forces of vehicle motion are linear functions of the vehicle mass (Sullivan & Cobas-Flores 2001, IFEU 2003). Once this relationship is established, the total GHG emissions from the vehicle use phase (including the fuel cycle) can be calculated for the mass-reduced vehicles, given that baseline fuel economy, mass savings, total vehicle mileage and GHG emissions from production, delivery and combustion of a unit of fuel are known.

Previous studies have shown that GHG emissions from vehicle end-of-life management operations, like shredding of vehicles, separation of the different metal and plastic fractions, and landfilling of automotive shredder residue (ASR), are very small relative to the other life cycle stages (Sullivan et al. 1998, Sullivan & Cobas-Flores 2001, DBJ 2004, Schmidt et al. 2004, DaimlerChrysler 2006). Any GHG emission differences due to different separation processes required by different automotive materials are thus also likely to be very small. The processes of vehicle shredding and material separation and landfilling are therefore not accounted for in the presented assessment methodology.

However, vehicle end-of-life management generates considerable amounts of economic output, mostly in the form of automotive end-of-life metal scrap. GHG emission allocation or system expansion due to end-of-life scrap generation at vehicle manufacturing is modelled explicitly in the presented methodology. The scrap input into a vehicle life cycle is generally considerably smaller than the output. Vehicles are thus net generators of scrap, the majority of which is end-of-life scrap. Automotive end-of-life scrap is typically recycled and used either in automotive or non-automotive applications. During scrap collection and processing, contamination with problematic substances and unsuitable mixing of alloys should be minimised, since this can impact the yield and quality of the secondary metal (Birat et al. 1999, Russo et al. 1999, EAA 2000, Utigard 2005).

2 A methodology for Life Cycle GHG Emission Assessments of Automotive Materials

2.1 General Aspects

This study was commissioned by WorldAutoSteel, the automotive group of the International Iron and Steel Institute (IISI). It was conducted by Roland Geyer, Assistant Professor at the Donald Bren School of Environmental Science and Management, which is part of the University of California at Santa Barbara (UCSB). The objective of this study is to develop a methodology for the parametric modelling and assessment of the life cycle GHG emissions related to automotive materials, with particular emphasis on body-in-white (BIW) designs based on mild steel, aluminium, and advanced high strength steels (AHSS), such as the Ultra Light Steel Auto Body - Advanced Vehicle Concept (ULSAB-AVC). For this purpose a parametric model has been developed which calculates life cycle GHG emissions attributable to vehicles as a function of their material composition and power train design. As explained in Section 1.2, the assessment methodology underlying the parametric model is based on life cycle assessment (LCA) according to ISO 14040/44 (2006). The scope of the life cycle impact assessment is limited to climate change impacts related to automotive material choice. Section 2.2 contains the definition of the goal of this study. In Section 2.3, functional unit and system boundaries are defined. Section 2.4 explains how the reference flows are derived from the definition of the functional unit. Section 2.5 details the parametric model of the inventory analysis, with particular emphasis on material production and recycling (Section 2.5.1) and vehicle use (Section 2.5.2). Section 2.6 briefly explains how impact assessment is integrated into the modelling methodology. Model transparency was a key criterion for this study. To facilitate peer-review, all model calculations and assumptions are presented explicitly and in detail. The implementation of the parametric model is described in Section 2.7. Section 2.8 details all the input data that are required to populate the model.

2.2 Goal of the Study

The immediate goal of this study is to develop a parametric model that quantifies the net changes in the GHG emissions that are attributable to passenger vehicles and that result from replacing a mild steel BIW with an aluminium-based or an AHSS-based BIW. To ensure meaningful comparisons, all three vehicles have to be functionally equivalent. The study uses LCA methodology as outlined in ISO 14040/14044 (2006). The life cycle inventory modelling is highly parameterised. Parametric life cycle modelling has been done previously (see e.g. Sullivan & Hu 1995) and has been chosen for two main reasons:

1. To separate the computational structure of the model from its input data.
2. To facilitate sensitivity analysis of the model results with respect to input data and modelling choices.

The broader goal of the study is to contribute to the scientific debate on the GHG implications of automotive material choice. For over a decade, studies containing comparative assertions disclosed to the public have been published on this issue. Due to differences in modelling methodology and input data, they are not in agreement. The parametric model developed in this study is designed to facilitate the building of consensus with regard to choosing the appropriate modelling methodology and selecting the required input data. The focus of this study is on computational structure, model sensitivity and data requirements. The model is aimed at supporting the application of LCA in the automotive industry to enhance environmental performance. The target audience includes everyone who has a stake in or is interested in this issue and is familiar with the basic principles of LCA and automotive engineering.

2.3 Scope of the Study

2.3.1 Functional Unit and Reference Flows

The primary function of the product system studied herein, cars and trucks, is to provide people with personal mobility. The study is thus concerned with regular passenger cars and excludes motorcycles and freight vehicles, like vans and trucks. Secondary functions such as safety, comfort and status are assumed to be approximately equal for all reference flows derived from the functional unit. The functional unit FU of the study is defined as follows:

FU : Transportation services of passenger cars of equivalent size, utility, equipment and power train configuration over their total vehicle life.

The resulting reference flows are three passenger cars of equivalent size, utility, equipment and power train configuration but with BIWs made from different materials. BIW designs based on

AHSS and aluminium are both treated as ways to reduce the mass of vehicles whose BIW is based on mild steel. To explicitly document the mass savings achieved by AHSS- and aluminium-intensive BIW designs, a conventionally designed baseline vehicle is defined in terms of its vehicle mass, material composition, power train design, and the resulting fuel economy. This is the first reference flow. AHSS- and aluminium-intensive design changes are then specified and used to derive material composition and fuel economy of the other two reference flows.

2.3.2 System Boundaries

Vehicle life cycles are very complex product systems (Keoleian et al. 1998). In this methodology, boundaries are chosen based on the objective to assess the differences in life cycle GHG emissions between the three reference flows, rather than to comprehensively capture all life cycle GHG emissions of vehicles (see Figure 5). All included processes are shown in black boxes with solid lines. They are primary and secondary production of steel (mild and AHSS) and aluminium ingots and slabs, further processing of these ingots and slabs into finished material products, vehicle use, and fuel production and delivery. Further processing includes rolling, galvanising, extruding and shape casting. It excludes all other forming operations, such as forging and stamping, and all joining operations. The main process groups that are excluded are shown in grey boxes with dashed lines. They are production and finishing of all materials other than steel and aluminium, vehicle manufacturing, and vehicle end-of-life management. Considering the contribution analyses shown in Figure 1 and Figure 2, and assuming that around 28% of the mass of a passenger vehicle comes from materials other than steel and aluminium, this boundary choice captures roughly between 90% and 95% of the life cycle GHG emissions of passenger cars.

However, the primary goal of the assessment methodology is to quantify GHG emission differences between vehicles that only differ in their BIW materials and the resulting secondary mass savings. This justifies the omission of all processes that are not or not significantly affected by the material choice for the BIW. Examples for such processes are production of manufacturing equipment and transportation infrastructure, distribution of materials, components and the finished vehicle. For example, transportation of a new vehicle from the manufacturing plant to the dealer has been estimated to account for at most half a percent of total life cycle GHG emissions of passenger vehicles (Bren School 2005). The GHG savings from transporting a lighter vehicle from plant to dealership will thus be at best a few per mille of total life cycle GHG emissions of the vehicle. Vehicle end-of-life operations (without the impact of scrap generation) is estimated to account for one or a few per mille of total life cycle GHG emissions of passenger vehicles (Sullivan et al. 1998, DBJ 2004). It is thus reasonable to assume that the difference in GHG emissions from end-of-life management due to different material composition of the vehicles will also be no more than one or a few per mille of total life cycle GHG emissions. The materials that are outside of the system boundaries are predominantly plastics, rubber, glass, and some non-ferrous base metals such as copper, zinc, nickel and lead. These materials are only indirectly affected through the secondary mass savings made possible through the mass reduction of the BIW. However, the vast majority of secondary mass savings will be in steel and aluminium. Assuming that all secondary mass savings are in steel and aluminium is thus estimated to generate an insignificant error.

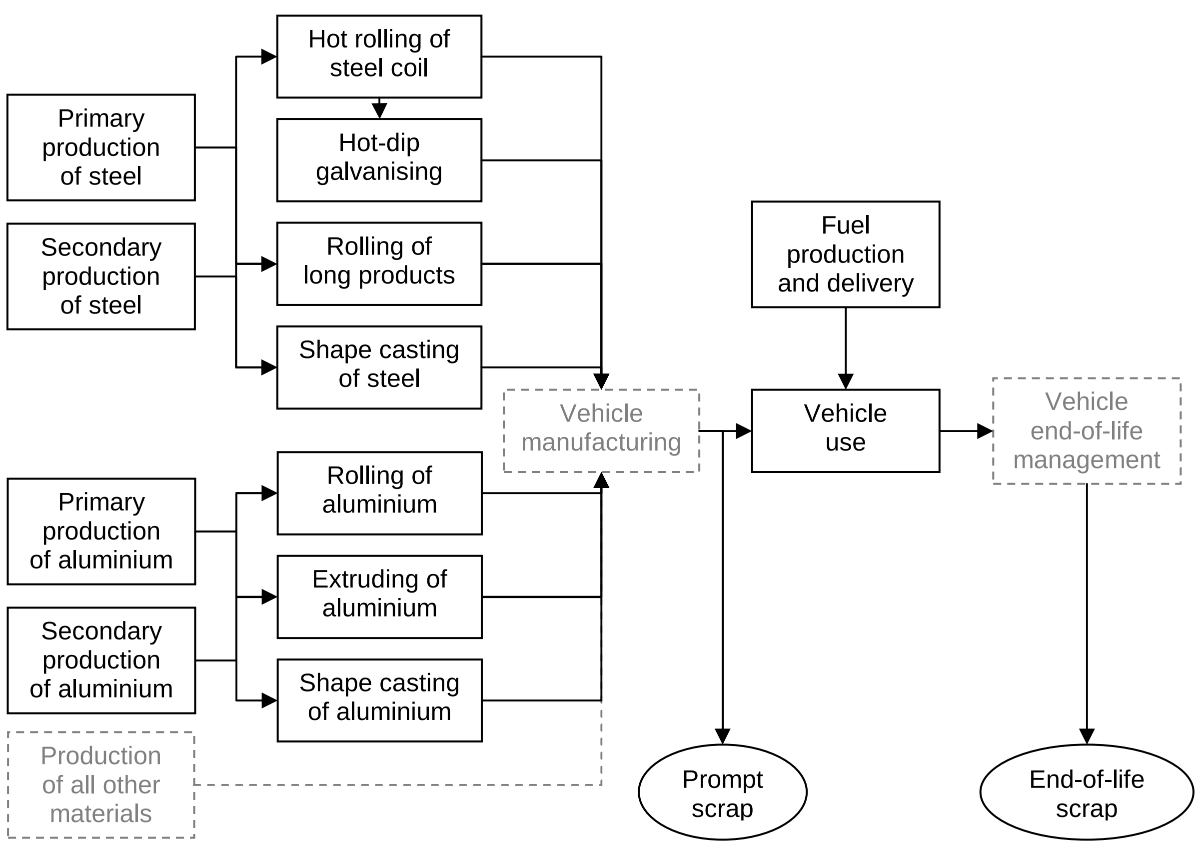

Figure 5: System boundaries of the study. Included processes are in black boxes with solid lines, excluded processes in grey boxes with dashed lines, ovals signify product flows leaving the vehicle life cycle.

Vehicle manufacturing (without the impact of scrap generation) is estimated to account for 4 to 7 percent of total life cycle GHG emissions of passenger cars (Sullivan et al. 1998, Sullivan & Cobas-Flores 2001, DBJ 2004, Bren School 2005). GHG emission differences due to the different manufacturing processes required for the different BIW materials might thus conceivably be in the order of 1 percent of total life cycle GHG emissions. Previous studies that estimate GHG emission differences from manufacturing steel and aluminium BIWs are considerably lower, however, and for this reason all vehicle manufacturing processes have been omitted from this study (Ecobilan 1997). However, this is estimated to be the largest error source in the methodology and thus should be the first system boundary issue addressed in any future work.

2.4 Parametric Model of Reference Flows

The functional unit FU is translated into three functionally equivalent reference flows RF^y $y = b, a, u$. The three reference flows describe vehicles of the same vehicle class, vehicle mass and power train configuration, but with different body-in-whites (BIW):

- RF^b : Vehicle with a BIW made entirely of mild steel
- RF^a : Vehicle with a BIW made entirely of aluminium
- RF^u : Vehicle with a BIW made entirely of AHSS

For the modelling purposes of the study, the three reference vehicles are fully characterised by their fuel economy, FE^y, and a 7-dimensional material composition vector, m_i^y:

$$RF^y = \left(m_1^y, m_2^y, m_3^y, m_4^y, m_5^y, m_6^y, m_7^y, FE^y\right), \qquad y = b, a, u \tag{1}$$

2.4.1 Material Composition of the Reference Vehicles

The material composition of reference flow RF^b (vehicle with mild steel BIW) is characterised by its total mass VW^b (given in kg) and the mass percentages of three steel and three aluminium categories, mc_i^b. The material categories are: flat carbon steel $(i = 1)$, long and special steel $(i = 2)$, cast steel $(i = 3)$, rolled aluminium $(i = 4)$, extruded aluminium $(i = 5)$, cast aluminium $(i = 6)$. The mass of each material category contained in the vehicle, m_i^b, is calculated as total mass times mass percentage $m_i^b = VW^b \cdot mc_i^b$. The remainder of the vehicle mass is assigned to a generic seventh category $(i=7)$ containing all other materials, $m_7^b = VW^b - \sum_{i=1}^{6} m_i^b$. The result is a material composition vector containing seven material categories, $m_i^b \quad i=1,...,7$, with $VW^b = \sum_{i=1}^{7} m_i^b$. The change in material composition that results from the AHSS- and aluminium-intensive designs of reference flows RF^z, $z=a,u$, is characterised by the following set of parameters:

ΔM	total mass of replaced material
k^z	material replacement coefficient of vehicle design z $(z=a,u)$
s	ratio of secondary mass savings to primary mass savings
π_i	material composition of replaced material $\left(\sum_{i=1}^{7} \pi_i = 1, \pi_4 = \pi_5 = \pi_6 = \pi_7 = 0\right)$
$\rho_\mathrm{i}^\mathrm{z}$	composition of replacing material of vehicle design z $\left(\sum_{i=1}^{7} \rho_\mathrm{i}^\mathrm{z} = 1, z=a,u, \rho_1^\mathrm{a} = \rho_2^\mathrm{a} = \rho_3^\mathrm{a} = \rho_7^\mathrm{a} = 0, \rho_4^\mathrm{u} = \rho_5^\mathrm{u} = \rho_6^\mathrm{u} = \rho_7^\mathrm{u} = 0\right)$
σ_i	material composition of secondary mass savings $\left(\sum_{i=1}^{7} \sigma_i = 1, \sigma_7 = 0\right)$

Material composition vectors of the AHSS- and aluminium-intensive designs are calculated as

$$m_i^z = m_i^b - \pi_i \Delta M + \rho_i^z k^z \Delta M - \sigma_i s(1-k^z)\Delta M \quad \text{with } z=a,u . \tag{2}$$

Resulting vehicle mass of the AHSS- and aluminium-intensive designs are calculated as

$$VW^z = \sum_{i=1}^{7} m_i^z = VW^b - \Delta M + k^z \Delta M - s(1-k^z)\Delta M . \tag{3}$$

Net weight savings of the two lightweight designs are therefore

$$\Delta VW^z = VW^b - VW^z = (1+s)(1-k^z)\Delta M . \tag{4}$$

The following example is only meant to illustrate the computational approach and was developed together with experts from the IISI WorldAutoSteel working group:

$VW^b = 1260kg$	$\Delta M = BIW^b = 360kg$
$mc_i^b = (0.4,0.15,0.1,0.01,0.01,0.05,0.28)$	
$k^a = 0.6$	$k^u = 0.75$
$\pi_i = (0.9,0.1,0,0,0,0,0)$	
$\rho_\mathrm{i}^\mathrm{a} = (0,0,0,0,0.7,0.3,0)$	$s = 0.3$
$\rho_\mathrm{i}^\mathrm{u} = (0.9,0.1,0,0,0,0,0)$	$\sigma_\mathrm{i} = (0.3,0.2,0.0,0.1,0.1,0.3)$

The material composition of reference flow RF^b simply follows from $m_i^b = VW^b \cdot mc_i^b$. The material compositions of reference flows RF^z, $z = a,u$, are derived from RF^b with the help of equation (2). The results are shown in Table 1.

(All values in kg)	i	mc_i^b	m_i^b	m_i^a	m_i^u
Flat carbon steel	1	40%	504.0	167.0	414.9
Long and special steel	2	15%	189.0	144.4	174.6
Cast steel	3	10%	126.0	126.0	126.0
Rolled aluminium	4	1%	12.6	159.5	9.9
Extruded aluminium	5	1%	12.6	73.1	9.9
Cast aluminium	6	5%	63.0	50.0	54.9
All other material	7	28%	352.8	352.8	352.8
Total weight		100%	1260.0	1072.8	1143.0

Table 1: Example calculation of the material compositions of the three reference vehicles

The parametric model of the material composition of the reference vehicle is descriptive rather than normative. This means that it does not attempt to predict the material composition of the reference vehicles based on their design characteristics but rather describes it based on empirical information gathered from industry and academia. The mass of the baseline vehicle depends, among other things, on the vehicle class and the power train configuration. The impact of those two vehicle characteristics should be studied via sensitivity analysis.

2.4.2 Fuel Economy of the Reference Vehicles

The characterisation of the reference flows is completed by determining the fuel economy of the vehicles. The average fuel economy of the reference flow RF^b (vehicle with mild steel BIW) is denoted by the parameter FE^b (given in $litres/100km$). The value of the fuel economy not only depends on vehicle mass and power train configuration but also on the type of fuel used and the driving characteristics, which are typically specified through so-called driving cycles. The impact of those four vehicle characteristics should be studied via sensitivity analysis.

The fuel economy of the reference flows corresponding to the two lightweight designs RF^z, $z = a,u$, is derived from the baseline fuel economy by subtracting the fuel economy improvement ΔFE^z generated by the mass savings, i.e.

$$FE^z = FE^b - \Delta FE^z, \; z = a,u . \tag{5}$$

The fuel economy improvement ΔFE^z of the two lightweight designs is calculated as

$$\Delta FE^z = FS \cdot \Delta VW^z,\ z = a,u, \tag{6}$$

where FS denotes the ratio of fuel savings per mass savings in $litre/(100km \cdot 100kg)$. Using equation (4), the fuel economy improvement can be expressed as a function of replaced mass:

$$\Delta FE^z = FS(1+s)(1-k^z)\Delta M^z,\ z = a,u \tag{7}$$

In literature, mass-reduction related fuel economy improvements are often calculated using the concept of a mass elasticity of the fuel economy, i.e. the ratio of relative fuel savings per relative mass savings $\frac{\Delta FE \cdot VW}{FE \cdot \Delta VW}$. This ratio of percentage changes is typically assumed to be constant, e.g. 0.5, which means that mass savings of $X\%$ generate fuel savings of $0.5 \cdot X\%$. The use of the economic concept of elasticity is problematic for at least two reasons: First, a relative fuel economy improvement of $X\%$ yields different absolute fuel economies, depending on whether the unit of measurement used is *mpg* or $litres/100km$. Second, functions with constant elasticity are of the form $f(x) = a \cdot x^b$, i.e. generally non-linear. However, if $\frac{\Delta FE \cdot VW}{FE \cdot \Delta VW} = constant$, then the fuel economy is a linear function of the vehicle weight, which directly contradicts the assumption of a constant elasticity. These problems are avoided in this model since the fuel savings due to mass savings, FS, are given in absolute instead of relative terms as described above.

FS is a critical model parameter, which has to summarise a complex reality (Wallentowitz et al. 2000, An & Santani 2004). Separate research has been carried out by FKA (2005) to investigate how FS depends not only on vehicle mass reduction, but also on parameters like power train configuration, driving cycles and type of power train adjustment. The results of the FKA study clearly show that the assumption of a constant elasticity is not a good approximation. It also shows that the value of FS depends on vehicle mass, power train configuration, driving cycle and power train adjustment and thus has to be modelled as a function of these parameters. The impact of those four vehicle characteristics should be studied via sensitivity analysis. Parameters like rolling resistance coefficient, aerodynamic resistance coefficient, frontal area and energy demand of accessories (zero load) are assumed to be identical for all reference vehicles.

2.5 Parametric Model of Life Cycle Inventory Analysis

The parametric model of the inventory analysis is based on attributional LCA, which facilitates benchmarking with previous studies and also serves as a natural starting point for additional consequential modelling. In this methodology the inventory of the life cycle GHG emissions, $LC\vec{I}^{\,y}$, attributable to each reference flow, RF^{y} $\;y = b,a,u$, is calculated as the sum of the GHG emissions from all its life cycle stages:

$$LC\vec{I}^{\,y} = \vec{I}^{\,y}_{prod} + \vec{I}^{\,y}_{man} + \vec{I}^{\,y}_{use} + \vec{I}^{\,y}_{eol} \quad \text{with} \quad y = b,a,u \tag{8}$$

$\vec{I}^{\,y}_{prod}$ are the cradle-to-gate emissions from the material production stage of vehicle *y*, $\vec{I}^{\,y}_{man}$ are the emissions from the vehicle manufacturing stage (including prompt scrap recycling), $\vec{I}^{\,y}_{use}$ the emissions from the vehicle use phase (including emissions from fuel production and delivery), and $\vec{I}^{\,y}_{eol}$ the emissions from its end-of-life (eol) management (including eol scrap recycling).

At the vehicle manufacturing and eol stages, the inventory model currently only covers the impact of co-producing prompt and eol scrap, i.e. the emissions from the manufacturing and eol management processes are not accounted for. Scrap recycling is included since many previous studies show that the GHG implications of prompt and end-of-life scrap recycling are significant and need to be accounted for (Saur et al. 1995, Stodolski et al. 1995, Ecobilan 1997, Takamatsu and Ohashi 2000, IAI 2000, Das 2000, Field et al. 2000, Dhingra et al. 2001, Hayashi et al. 2001, Birat et al. 2004). The scrap flows into and out of the vehicle life cycle create the need for allocation or system expansion in attributional LCA. The inventory model quantifies the GHG impacts from material production and recycling in conjunction. As a result of this and the above-mentioned process omissions, equation (8) simplifies to:

$$LC\vec{I}^{\,y} = \vec{I}^{\,y}_{mat} + \vec{I}^{\,y}_{use} = \left(\vec{I}^{\,y}_{prod} + \vec{I}^{\,y}_{prompt-scrap} + \vec{I}^{\,y}_{eol-scrap}\right) + \vec{I}^{\,y}_{use} \quad \text{with} \quad y = b,a,u \tag{9}$$

$LC\vec{I}^{\,y}$ does not calculate the total amount of GHG emissions attributable to a vehicle life cycle, but rather the GHG emissions related to production and recycling of automotive steel and aluminium and the use of the resulting vehicles. This narrowed scope is deemed appropriate for the goal of the study, the calculation of life cycle GHG emission differences between the reference flows, and therefore considered in agreement with ISO 14040/44 (2006).

2.5.1 Inventory Model of Material Production and Recycling

For each reference flow RF^y, $y = b, a, u$, the GHG emission inventory of material production and recycling is calculated as

$$\vec{I}^{\,y}_{mat} = \sum_{i=1}^{6} \frac{m_i^y}{\gamma_i} \vec{I}_i^{\,att} \quad y = b, a, u, \tag{10}$$

where $\vec{I}_i^{\,att}$ are the GHG emissions attributable to one kg of material category *i* used in vehicle manufacturing. For each material type, the mass required for vehicle manufacturing, i.e. the shipped material, is the material content divided by the manufacturing yield, i.e. m_i^y / γ_i, where γ_i denotes the manufacturing yield of material category *i*. Note that the model does currently not account for GHG emissions from automotive materials other than steel and aluminium. This is considered to be in agreement with the scope of the study, which is to quantify GHG emission differences between mild-steel-, AHSS- and aluminium-intensive vehicle designs rather than to estimate total life cycle emissions of vehicles.

The main challenge posed by equation (10) is the calculation of $\vec{I}_i^{\,att}$, which essentially deals with the question of how to account for the use and generation of scrap and the resulting metal products from secondary production (see e.g. Ecobilan 1997, Schmidt et al. 2004) . The secondary production routes of steel and aluminium have much lower GHG emissions than their respective primary production routes. In the case of aluminium, GHG emissions per kg of ingot differ by a factor of about 20. It is thus of utmost importance to make sound and consistent choices for $\vec{I}_i^{\,att}$. There are many possibilities of allocating elementary flows in product systems with recycling (see e.g. Klöpfer 1996, Ekvall & Tillman 1997). One possibility is to simply track which of the materials used in the vehicles come from primary and which from secondary production routes, and multiply them with their respective cradle-to-gate GHG emissions (see e.g. Ecobilan 1997, IAI 2000, Dhingra et al. 2001, Hayashi et al. 2001). This approach accounts for the GHG emissions generated during the production of the automotive materials used in the vehicle life cycle. However, it does not account for any GHG impacts related to the consumption and generation of metal scrap, which has been the main criticism of this so-called secondary content or cut-off method (see e.g. Ekvall & Tillman 1997, Atherton 2007). The USAMP Generic

Vehicle Life Cycle Inventory Study (Sullivan et al. 1998), for example, accounts for prompt scrap contribution to the LCI. Details on its methodology can be found in Keoleian et al. (1998).

One way to account for scrap recycling at the vehicle manufacturing and end-of-life management stages is the so-called avoided burden approach, or consequential system expansion, as described in Section 1.2 (Guinée 2002, Frischknecht 2006). Here the system boundaries are expanded to include the secondary production processes that use the prompt and end-of-life scrap as inputs and the changes in other production processes, e.g. primary production of the same material type, that result from the increase in scrap recycling. The avoided burden approach models changes in flows and processes and is thus based on consequential reasoning. In the standard use of this approach for metals it is assumed that scrap recycling reduces primary production of the same metal by an equal amount (Stodolsky et al. 1995, Ecobilan 1997, Das 2000, Field et al. 2000, Brimacombe et al. 2001). Typically, no attempt is made to verify this assumption. In the standard use for metals, the avoided burden approach thus gives a scrap-generating product system an emission credit that is equal to the emission difference between primary and secondary production multiplied by the amount of metal produced from scrap recycling.

In attributional LCA, the sum of the life cycle inventories of two product systems has to be equal to the life cycle inventory of the joint product system (Ekvall & Tillman 1997, Guinée 2002). Use of the avoided burden approach thus requires that product systems that use metal from secondary production receive an emission debit of the same size as the emission credit given to product systems that generate scrap for secondary production. In the standard use for metals, this is equal to the emission difference between primary and secondary production multiplied by the amount of secondary content. From a perspective of consequential system expansion, this is equivalent to assuming that increased recycled content only diverts scrap from other applications instead of increasing scrap collection and recycling. Atherton (2007) e.g. argues that “the specific origin of input material (whether recycled or primary) is not relevant […]” since, for metals, specifying recycled content is ineffective for reducing environmental impact. Typically, no attempt is made to verify this assumption. From an attributional perspective, the avoided burden approach can also be regarded as a method to calculate a GHG emission inventory for scrap, which is thus treated as a product flow rather than an elementary waste flow (IISI 2005).

A large group of metal industry associations, including IAI and IISI, recently expressed their strong support of the avoided burden approach (Atherton 2007). It has been pointed out in literature that cut-off and avoided burden approach are the two extreme cases of a possible continuum of credit/debit schemes (Ekvall 2000, UBA 2002, Ekvall & Weidema 2004), with the so-called 50/50 method in the centre (Klöpfer 1996, Ekvall & Tillman 1997, UBA 2002). Without loss of generality, the presented methodology thus uses a sliding credit/debit system (CDS) with the fraction of recycling credit given to the vehicle life cycle as a model parameter. It thus supports an infinite number of allocation methods, including cut-off, avoided burden and 50/50, which facilitates sensitivity analysis. From a consequential perspective, the sliding credit/debit system can be regarded as a refined way of applying system expansion. From an attributional point of view, it can be interpreted as a refined way of calculating a GHG inventory for scrap.

In its own publications, the International Iron and Steel Institute (IISI) endorsed two allocation methods, the avoided burden approach and the so-called multi-step recycling method (IISI 2005, Birat et al. 2005). The multi-step approach has been suggested earlier in literature (Borg & Anderson 1998). The standard use of the avoided burden approach, or consequential system expansion, for metals is based on the assumption that use, collection, separation and recycling of the metal leave its inherent properties unchanged, which thus makes it potentially recyclable for an infinite number of times. However, the avoided burden approach accounts only for one recycling cycle. The multi-step recycling method is designed to reflect not only the recycling of the prompt and end-of-life scrap from the vehicle life cycle but also the subsequent cycles of the metal. Clause 4.3.4.3.4 of ISO 14044 (2006) mentions the number of subsequent uses of a recycled material as a basis for allocation, but does not give it a high priority. In order to facilitate sensitivity analysis of the model results with respect to allocation/system expansion, the multi-step recycling (MSR) method is included in the presented methodology. In the limit case of infinite recycling cycles and identical recycling rates for all cycles, the MSR method yields results identical to the standard avoided burden approach for metals. However, the two methods are based on fundamentally different allocation principles. All allocation methods supported by the presented methodology have been chosen for their suitability to account for recycling of automotive metals. They are thus not necessarily equally appropriate for other automotive materials.

2.5.1.1 Allocation via sliding credit/debit system (CDS)

The emissions attributable to each material type are calculated as follows:

$$\vec{I}_i^{att} = \left(1 - r_i^{cont}\right)\vec{I}_i^{\,p} + r_i^{cont}\vec{I}_i^{\,s} + \vec{I}_i^{\,f} - \alpha\left(\frac{s_i^{out} - s_i^{in}}{s_i^s - s_i^p}\right)\left(\vec{I}_i^{\,p} - \vec{I}_i^{\,s}\right) \tag{11}$$

The maximum amount of displaced primary production, $(s_i^{out} - s_i^{in})/(s_i^s - s_i^p)$, which determines the maximum recycling credit, is derived on the next page by balancing all external scrap flows (see Figure 6). The definitions of all the necessary parameters are:

$\vec{I}_i^{\,p}$	ore/scrap - to - ingot emissions (primary production route) of 1 *kg* of material type *i*
$\vec{I}_i^{\,s}$	scrap - to - ingot emissions (secondary production route) of 1 *kg* of material type *i*
$\vec{I}_i^{\,f}$	ingot - to - finished - material emissions (further processing) of 1 *kg* of material type *i*
r_i^{cont}	percentage of shipped material type *i* coming from the secondary production route
s_i^s	kg of scrap input per kg of slab/ingot from the secondary production route
s_i^p	kg of scrap input per kg of slab/ingot from the primary production route
s_i^{in}	gross amount of scrap input (in *kg*) into the vehicle life cycle
s_i^{out}	gross amount of collected and separated scrap output (in *kg*) from the vehicle life cycle
α	$\in [0,1]$, amount of recycling credit allocated to the vehicle life cycle

The scrap flows in and out of the vehicle life cycle and the combined recycling rate of prompt and end-of-life vehicle scrap are calculated as

$$\begin{aligned} s_i^{in} &= s_i^p(1 - r_i^{cont}) + s_i^s r_i^{cont} \\ s_i^{out} &= ce_i^m se_i^m (1 - \gamma_i) + ce_i^{eol} se_i^{eol} \gamma_i \\ r^{car} &= ce_i^m se_i^m ry_i^m (1 - \gamma_i) + ce_i^{eol} se_i^{eol} ry_i^{eol} \gamma_i \end{aligned} \quad . \tag{12}$$

The definitions of the additional parameters are

ce_i^m	collection efficiency of automotive manufacturing (prompt) scrap of material type *i*
se_i^m	separation efficiency of automotive manufacturing (prompt) scrap of material type *i*
ry_i^m	recycling yield of automotive manufacturing (prompt) scrap of material type *i*
ce_i^{eol}	collection efficiency of automotive end - of - life (eol) scrap of material type *i*
se_i^{eol}	separation efficiency of automotive end - of - life (eol) scrap of material type *i*
ry_i^{eol}	recycling yield of automotive end - of - life (eol) scrap of material type *i*
γ_i	manufacturing yield of material type *i*

Scrap recycling rates are defined as secondary material output over scrap generation and thus account for collection efficiency, separation efficiency and the metal yield of secondary production. The average scrap input to secondary production is calculated as $s_i^s = s_i^{out} / r_i^{car}$.

For $\alpha = 1$, equation (11) gives maximum credit to the generation of recycled scrap and maximum debit to the use of scrap. The value for this maximum credit/debit is calculated by balancing all the scrap flows in Figure 6 and assuming that any changes in scrap input or output cause displacement between secondary and primary production outside of the vehicle life cycle as assumed by the standard avoided burden approach for metals:

$$s_i^{out} + s_i^p \beta = s_i^{in} + s_i^s \beta \Rightarrow \beta = \frac{s_i^{out} - s_i^{in}}{s_i^s - s_i^p} \tag{13}$$

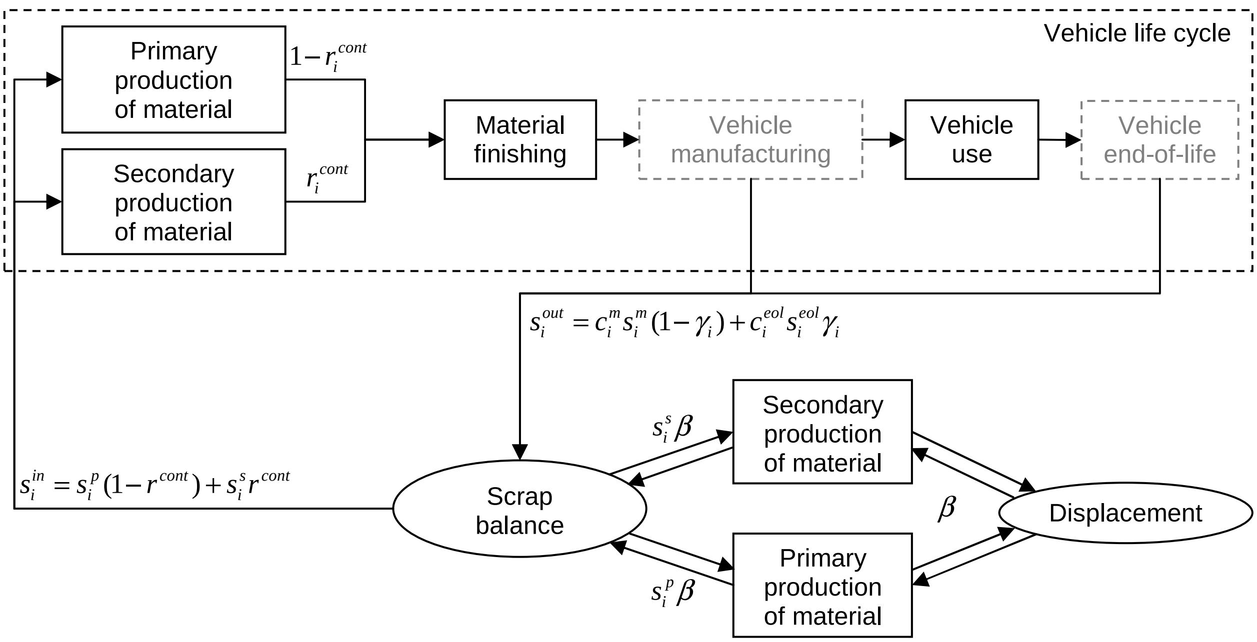

Figure 6: The standard avoided burden approach for maximum credit/debit calculation

Figure 6 shows that the avoided burden approach used for the credit/debit system accounts for scrap consumption of primary metal production, as is the case in BF/BOF steel production. In the case that $s_i^p = 0$, it follows that $\beta = r_i^{car} - r_i^{cont}$. For $\alpha = 1$, equation (11) now simplifies to

$$\vec{I}_i^{att} = \vec{I}_i^p + \vec{I}_i^f - r_i^{car}\left(\vec{I}_i^p - \vec{I}_i^s\right) = \left(1 - r_i^{car}\right)\vec{I}_i^p + r_i^{car}\vec{I}_i^s + \vec{I}_i^f \tag{14}$$

and no longer reflects the use of material from secondary production, i.e. is independent of r_i^{cont}.

For $\alpha = 0$, equation (11) simplifies to

$$\vec{I}_i^{att} = \left(1 - r_i^{cont}\right)\vec{I}_i^{\,p} + r_i^{cont}\vec{I}_i^{\,s} + \vec{I}_i^{\,f} \tag{15}$$

This is the previously discussed recycled content or cut-off method, which only accounts for the emissions that occur during production and finishing of the material contained in the vehicle. The same is true if $s_i^{out} = s_i^{in}$, i.e. if the vehicle uses and generates identical amounts of scrap.

In the case of steel, primary production emissions are based on BF/BOF technology and secondary production emissions are based on EAF technology. Primary steel thus denotes steel from the BF/BOF or primary production route, which can contain significant amounts of scrap. A distinction thus needs to be made between the secondary route content of a vehicle r_i^{cont} and the scrap content of a vehicle. The latter is not a straightforward concept, since it is unclear how to determine the scrap yield of primary steel production. One possible definition of automotive scrap content is $s_i^{in} / s_i^s = (1 - r_i^{cont}) s_i^p / s_i^s + r_i^{cont}$. In the case of aluminium, primary production emissions are based on Hall-Heroult electrolytic process technology and secondary production emissions on secondary remelting and refining technology.

2.5.1.2 Allocation via multi-step recycling method (MSR)

Here, the emissions attributable to each material type are calculated as follows:

$$\vec{I}_i^{att} = \left(1 - r_i\right)\vec{I}_i^{\,p} + r_i\vec{I}_i^{\,s} + \vec{I}_i^{\,f} = \vec{I}_i^{\,p} + \vec{I}_i^{\,f} - r_i\left(\vec{I}_i^{\,p} - \vec{I}_i^{\,s}\right) \tag{16}$$

$$\text{with } r_i := \frac{r_i^{car}\sum_{m=0}^{n-1}\left(r_i^{all}\right)^m}{1 + r_i^{car}\sum_{m=0}^{n-1}\left(r_i^{all}\right)^m} = \frac{r_i^{car} - r_i^{car}\cdot\left(r_i^{all}\right)^n}{1 - r_i^{all} + r_i^{car} - r_i^{car}\cdot\left(r_i^{all}\right)^n} \tag{17}$$

Here are the definitions of all parameters used in equations (16) and (17):

n	number of recycling cycles
$\vec{I}_i^{\,p}$	cradle - to - ingot emissions (primary production route) of $1\,kg$ of material type i
$\vec{I}_i^{\,s}$	cradle - to - ingot emissions (secondary production route) of $1\,kg$ of material type i
$\vec{I}_i^{\,f}$	ingot - to - finished - material emissions (further processing) of $1\,kg$ of material type i
r_i^{car}	overall automotive recycling rate of material type i, i.e. $r_i^{car} := \left(1 - \gamma_i\right)r_i^{prompt} + \gamma_i r_i^{eol}$
r_i^{all}	overall recycling rate of material type i (for all its applications)

r_i is the overall recycling rate of primary route material of category *i*, that is recycled once with the automotive recycling rate, r_i^{car}, and then recycled another *n-1* times with the average recycling rate of material category *i* for all its applications, r_i^{all}.

In the case that $r_i^{all} = r_i^{car}$ the calculation of r_i simplifies to

$$r_i := \frac{r_i^{car} - \left(r_i^{car}\right)^{n+1}}{1 - \left(r_i^{car}\right)^{n+1}} \tag{18}$$

For $n \to \infty$, equation (18) further simplifies to $r_i = r_i^{car}$, given that $r_i^{car} < 1$. In this special case, the avoided burden approach of equation (14) and multi-step recycling yield the same amount of GHG emissions attributable to the production and recycling of material category *i*:

$$\vec{I}_i^{att} = \left(1 - r_i^{car}\right)\vec{I}_i^{p} + r_i^{car}\vec{I}_i^{s} + \vec{I}_i^{f}$$

If the overall recycling rate of the material is different from its automotive recycling rate, $r_i^{all} \neq r_i^{car}$, then the attributable emissions calculated based on avoided burden (with $s_i^p = 0$) will be different from those calculated based on multi-step recycling, even if $n \to \infty$:

$$\vec{I}_i^{p} + \vec{I}_i^{f} - r_i^{car}\left(\vec{I}_i^{p} - \vec{I}_i^{s}\right) \neq \vec{I}_i^{p} + \vec{I}_i^{f} - \frac{r_i^{car}}{1 - r_i^{all} + r_i^{car}}\left(\vec{I}_i^{p} - \vec{I}_i^{s}\right)$$

Avoided burden and multi-step recycling methods fundamentally differ in the way they allocate emissions to material production and recycling. The latter calculates the average emissions per kg of material for a metal production system consisting of initial primary production and *n* subsequent recycling cycles. Every kg of metal, regardless of recycled content or recycling fate, is attributed the same amount of emissions, the average of the specified metal production system. The avoided burden approach only accounts for the immediate scrap outputs and inputs from and to the product system and ignores the characteristics of the overall metal production system. Both methods, CDS and MSR, require the knowledge of a variety of recycling rates. It may be difficult to obtain individual prompt and end-of-life scrap recycling rates for the different steel and aluminium categories and thus more practical to combine the three steel grades $(i = 1,2,3)$ and the three aluminium grades $(i = 4,5,6)$ for the purpose of collecting or modelling prompt and end-of-life scrap recycling rates.

2.5.2 Inventory Model of Vehicle Use

For each reference flow RF^y, $y = b, a, u$, the GHG emission inventory of vehicle use is calculated as

$$\vec{I}^{\,y}_{use} = FE^y \cdot \vec{I}_{fuel} \cdot TM \quad , \ y = b, a, u \ , \tag{19}$$

The model parameters are defined as follows:

FE^y	Average fuel economy (in *litres/*100*km*) of reference vehicle *y* for given fuel type and driving cycles
$\vec{I}_{fuel}$	GHG emission inventory of production, delivery and combustion of 1 litre fuel
TM	total life of the vehicle (in *km*) (assumed to be the same for all three vehicle designs)

The use phase GHG emissions per kilometre are calculated as the product between vehicle fuel economy and the GHG intensity of fuel production, delivery and consumption. To illustrate the GHG emission impact of biofuel use, the inventory model can be adjusted to reflect the GHG emissions of vehicle use based on gasoline blended with ethanol made from three alternative types of fuel crops. The impact of ethanol content and biofuel crop should then be assessed via sensitivity analysis. Ethanol fuel crops are divided into starch-based such as corn (g), sugar-based such as sugar cane (s), or cellulose-based such as switch grass (c), since their production has very different GHG emission inventories (IEA 2004, Delucchi 2006). Adjusted GHG emissions per 100 driven kilometres, i.e. $FE^y \cdot \vec{I}_{fuel}$, for ethanol / gasoline blends are calculated as:

$$FE^y \cdot \vec{I}_{fuel} = FE^y_{gasoline} \cdot \vec{I}_{gasoline} \cdot \left(1 - ec \cdot ef_j\right) \tag{20}$$

$\vec{I}_{gasoline}$	GHG emission inventory of production, delivery and combustion of 1 litre gasoline
ec	Ethanol content (in volume %)
ef_j	Well - to - wheels GHG emission factor (in %) of ethanol from fuel crop $j = g, s, c$ relative to gasoline and per driven kilometre

The GHG emission factor of ethanol, ef_i, gives the well-to-wheels GHG emissions of ethanol as a percentage of the well-to-wheels GHG emissions of gasoline (given in $kgCO_2eq / km$) and thus summarises the assumptions for production, delivery and combustion of ethanol (IEA 2004, Delucchi 2006). This notation is used in most biofuel and hydrogen pathway analyses (see Delucchi (2006) for a recent review) and accounts for changes in volumetric ($liter/km$) and (MJ/km) calorific fuel economy that result from fuel changes and the resulting power train adjustments. Along the same lines, the presented methodology can be readily extended to model the GHG impacts of other fuel types, such as fossil diesel / biodiesel blends.

2.6 Life Cycle Impact Assessment

Impact assessment is the process of converting an inventory of elementary flows into a set of environmental impact indicators. In this study, the only considered elementary flows are GHG emissions, and the only considered impact indicator is climate change. The global warming potential GWP_x of a given inventory $\vec{I}_x$ of elementary flows is calculated as:

$$GWP_x = \sum_{l=1}^{m} gwp^l \cdot I_x^l \text{, with } \vec{I}_x = \left(I_x^1, I_x^2, I_x^3, ..., I_x^m\right)$$

The model parameters are defined as follows:

gwp^l	Global warming potential of greenhouse gas (GHG) l
I_x^l	Elementary flow of GHG l for process x
GWP_x	Global warming potential of process x for a given process inventory $\vec{I}_x$

Such a conversion of an inventory vector into an indicator result can be done on any level; process, sub-system, or whole system. In fact, it is possible to first convert the emission inventories of the unit processes from Figure 5 into indicator results and base all model calculations from Section 2.5 on the indicator results GWP_x (in kg CO2eq) rather than inventory data $\vec{I}_x$ of the processes. All equations in Section 2.5 remain the same, only with scalars of global warming potentials GWP_x instead of inventory vectors $\vec{I}_x$. This is, in fact, the way inventory analysis and impact assessment has been implemented in a spreadsheet version of the presented methodology. The sequence of inventory analysis and impact assessment is reversible since the inventory model contains only linear manipulations of the emission inventories of the unit processes. The list of greenhouse gases covered by the process inventories should be consistent across all processes and as comprehensive as possible. The impact assessment of the represented methodology could be readily extended to include more impact categories, given that all relevant elementary flows are covered by the process inventories and all relevant processes are contained within the chosen system boundaries. An extension of the impact assessment would thus require a re-examination of the available process inventories and the chosen system boundaries.

2.7 Implementation of Parametric Model

The presented methodology is readily translated into a parametric model. This section describes how the different elements of the parametric model have been implemented in a simple Excel-based spreadsheet model. Due to the parametric nature of the model, all input data can be modified. Choice of input data and model results are not part of the presented methodology and are discussed in a separate publication.

1. The GHG emission inventories of all processes within the system boundaries are converted into global warming potentials

   $GWP_x = \sum_{l=1}^{m} gwp^l \cdot I_x^l$

2. Material composition and fuel economy of the baseline reference flow RF^b are selected

   $RF^y = \left(m_1^b, m_2^b, m_3^b, m_4^b, m_5^b, m_6^b, m_7^b, FE^b\right)$

3. Material composition and total mass of the lightweight vehicle designs, RF^z, $z = a,u$, are calculated:

   $m_i^z = m_i^b - \pi_i \Delta M + \rho_i^z k^z \Delta M - \sigma_i s\left(1-k^z\right)\Delta M$ and

   $VW^z = \sum_{i=1}^{7} m_i^z = VW^b - (1+s)\left(1-k^z\right)\Delta M = VW^b - \Delta VW^z$

4. The gasoline-based fuel economies of the lightweight vehicle designs are calculated:

   $FE_{gasoline}^z = FE_{gasoline}^b - \Delta FE_{gasoline}^z$ with $\Delta FE_{gasoline}^z = FS \cdot \Delta VW = FS(1+s)\left(1-k^z\right)\Delta M$

5. Global warming potentials of the use phase are calculated for all three vehicles:

$$GWP_{use}^{y} = FE^{y} \cdot GWP_{fuel} \cdot TM \text{ with } FE^{y} \cdot GWP_{fuel} = FE_{gasoline}^{y} \cdot GWP_{gasoline}\left(1 - ec \cdot ef_{j}\right)$$

6. Attributable global warming potentials are calculated for each material category:

CDS: $GWP_i^{att} = \left(1 - r_i^{cont}\right)GWP_i^{p} + r_i^{cont} GWP_i^{s} + GWP^{f} - \alpha\left(\frac{s_i^{out} - s_i^{in}}{s_i^{s} - s_i^{p}}\right)\left(GWP_i^{p} - GWP_i^{s}\right)$ or

MSR: $GWP_i^{att} = (1 - r_i)GWP_i^{p} + r_i GWP_i^{s} + GWP^{f}$

7. Global warming potentials from material production and recycling are calculated for all three vehicles:

$$GWP_{mat}^{y} = \sum_{i=1}^{7} \frac{m_i^{y}}{\gamma_i} GWP_i^{att}$$

8. Global warming potentials of the product system are calculated for all three vehicles:

$$GWP^{y} = GWP_{mat}^{y} + GWP_{use}^{y}$$

9. The life cycle GHG emission differences between vehicles are calculated:

Baseline vs. Aluminium: $\Delta GWP^{b-a} = GWP^{b} - GWP^{a}$

Baseline vs. AHSS: $\Delta GWP^{b-u} = GWP^{b} - GWP^{u}$

AHSS vs. Aluminium: $\Delta GWP^{u-a} = GWP^{u} - GWP^{a}$

10. The crossover distances are calculated:

$$M_{X-over}^{b-a} = \frac{GWP_{mat}^{a} - GWP_{mat}^{b}}{\left(FE^{b} - FE^{a}\right)GWP_{fuel}} \qquad M_{X-over}^{u-a} = \frac{GWP_{mat}^{a} - GWP_{mat}^{u}}{\left(FE^{u} - FE^{a}\right)GWP_{fuel}}$$

The crossover distance between two reference flows is defined as the total driving distance at which both reference vehicles have the same amount of attributable life cycle GHG emissions. The difference between the cross over distance and the assumed vehicle life (193,080 km) indicates which reference vehicle has lower attributable life cycle GHG emissions.

2.8 Required Input Data

This section lists all the input data that are required to populate the parametric model. It is assumed that impact assessment is conducted at the process level, not the product system level. The required input data are thus the global warming potentials (per unit output) of the processes, rather than their process inventories. It is further assumed that recycling rates are only available for steel and aluminium as a whole and not for the individual steel and aluminium categories. However, the model can accommodate individual rates if they are available.

	VW^b (ICEV)	VW^b (HEV)	Replaced mass (BIW) ΔM
Compact			
Midsize			
SUV			

Table 2: Mass of reference flow RF^b and its BIW (in kg) as a function of vehicle size and power train configuration [1)]

$FE^b_{gasoline}$	ICEV (NEDC)	ICEV (Hyzem)	HEV (NEDC)	HEV (Hyzem)
Compact				
Midsize				
SUV				

Table 3: Gasoline-based fuel economy $FE^b_{gasoline}$ of reference flow RF^b (in litre/100km) as a function of vehicle size and power train configuration [2)]

FS	ICEV (NEDC)	ICEV (Hyzem)	HEV (NEDC)	HEV (Hyzem)
Compact				
Midsize				
SUV				

Table 4: Fuel savings per mass savings FS (in litres/100km and 100kg) without power train adjustment

FS	ICEV (NEDC)	ICEV (Hyzem)	HEV (NEDC)	HEV (Hyzem)
Compact				
Midsize				
SUV				

Table 5: Fuel savings per mass savings FS (in litres/100km and 100kg) with power train adjustment [3)]

[1)] ICEV = internal combustion engine vehicle, HEV = hybrid electric vehicle
[2)] NEDC = new European driving cycle (modal), Hyzem (transient)
[3)] Engine power is reduced to yield the same 0-100km/h acceleration as the baseline vehicle

	Composition of reference flow RF^b, mc_i^b	Composition of replaced material, π_i	Composition of replacing material		Composition of secondary mass savings, σ_i
			AHSS, ρ_i^u	Aluminium, ρ_i^a	
Flat carbon steel					
Long & special steel					
Cast steel					
Rolled aluminium					
Extruded aluminium					
Cast aluminium					

Table 6: Data required to calculate material compositions of reference flows $RF^y \ y = b, a, u$

Material replacement coefficient AHSS, k^u	%	
Material replacement coefficient aluminium, k^a	%	
Secondary mass savings coefficient, s	%	

Table 7: Data required to calculate primary and secondary mass savings

Vehicle life (total mileage in km), TM	
Well-to-wheel GWP of gasoline (in kgCO$_2$eq/litre), $GWP_{gasoline}$	
Ethanol content (in volume %), ec	
GHG emission factor of ethanol, $ef_j \quad j = g, s, c$ (in %)	
from grain-based fuel crops (j=g)	
from sugar-based fuel crops (j=s)	
from cellulose-based fuel crops (j=c)	

Table 8: Data required to calculate global warming potential from vehicle use, GWP_{use}^y

Material product	System boundaries	Symbol	kgCO$_2$eq/kg
BF /BOF slab	Cradle-to-slab	$GWP_{1,2,3}^p$	
EAF slab	Scrap-to-slab	$GWP_{1,2,3}^s$	
Hot rolled coil (HRC)	Slab-to-finished-product	GWP_{1a}^f	
Hot dip galvanised (HDG)	Slab-to-finished-product	GWP_{1b}^f	
Long and special	Slab-to-finished-product	GWP_2^f	
Cast steel	Slab-to-finished-product	GWP_3^f	

Table 9: Global warming potentials of steel production and further processing

HRC vs. HDG steel sheet in automotive (in % vs. %)	
Scrap input to primary steel production (BF/BOF route) (in kg/kg) $sp_{1,2,3}$	

Table 10: Data required to calculate global warming potential from steel production

Material product	System boundaries	Symbol	$kgCO_2eq/kg$
Primary ingot	Cradle-to-ingot	$GWP^{p}_{4,5,6}$	
Secondary ingot	Scrap-to-ingot	$GWP^{s}_{4,5,6}$	
Rolled aluminium	Ingot-to-finished-product	GWP^{f}_{4}	
Extruded aluminium	Ingot-to-finished-product	GWP^{f}_{5}	
Cast aluminium	Ingot-to-finished-product	GWP^{f}_{6}	

Table 11: Global warming potentials of aluminium production and further processing

Scrap input to primary aluminium production (Hall-Herault route) (in kg/kg) $sp_{4,5,6}$	

Table 12: Data required to calculate global warming potential from aluminium production

	Yield in vehicle manufacturing γ_i	Vehicle's content of material from secondary production route r_i^{cont}
Flat carbon steel		
Long & special steel		
Cast steel		
Rolled aluminium		
Extruded aluminium		
Cast aluminium		

Table 13: Manufacturing yields (in fraction of shipped material) and secondary contents for passenger vehicles (in fraction of total content of material type))

	Prompt scrap recycling		Eol scrap recycling	
	Symbol	I%	Symbol	%
Collection efficiency	ce_i^m		ce_i^{eol}	
Separation efficiency	se_i^m		se_i^{eol}	
Recycling yield	ry_i^m		ry_i^{eol}	

Table 14: Automotive prompt and end-of-life (eol) scrap recycling data

	Steel	Aluminium
Recycling allocation parameter α		
Number of recycling cycles n		
Overall recycling rate r_i^{all}		

Table 15: Additional data required for allocation via CDS: α
Additional data required for allocation via MSR: n, r_i^{all}

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4 Appendix A: External Review Panel Report

The following review has been performed as a critical review by a panel of interested parties according to Clause 6.3 of ISO 14044 (2006).

4.1 The Review Process

The panel chairman, Dr. Atsushi Inaba, was selected by IISI. The chairman selected the other panel members, Dr. Greg Keoleian, Dr. Gerald Rebitzer, and Dr. John Sullivan. A draft document was submitted to the review panel on 22 January 2007. On 1 March 2007 and 4 May 2007 the external review panel met with Dr. Roland Geyer to discuss draft document and underlying study. A decision was made to not review input data and model results but only the research methodology. A methodology report was submitted to the review panel on 7 May 2007. After receipt of draft comments and conclusions from the review panel, Dr. Geyer submitted an amended draft methodology report on 2 August 2007. This draft formed the basis of the final comments and conclusions that are reported below. The review panel results and comments were approved by all review panel members and sent to Dr. Geyer on 27 October 2007. The final version of the methodology report contains eight more editorial changes request by the panel and an extension of the recycling methodology based on Comment 2) b) of Reviewer 1. Please note that the reviewers' page references refer to the draft methodology report from 2 August 2007, while the author's page references refer to this final version of 7 December 2007.

4.2 Results of the External Review

The methodology presented in this study is well developed and complies with the concept of the life cycle approach of products of ISO 14040(2006) and ISO 14044(2006), although ISO does not assume this kind of LCI study mainly focusing on the deviation of GHG emissions from the baseline presenting the functional unit using parameters.

The methodology presented is generally consistent with ISO 14040 and 14044. While a full LCA includes results with multiple impact categories, this study is limited to the methodology and focuses exclusively on greenhouse gas emissions. But ISO 14044(2006) and ISO 14044(2006) accept an LCI study focusing on limited emissions. As the author states, the methodology presented in this study can relatively easily be expanded to address other impact categories as well. In general, assumptions, data sources, and data analysis methods are well documented, as requested by ISO 14040(2006) and ISO 14044(2006).

In this paper state of the art methods for recycling allocation were presented including the sliding credit/debit system. It is important to point out that these approaches each suffer from several limitations. As many of these limitations were articulated by the author, it must be strongly emphasized that the recycling allocation methods could influence the overall life cycle greenhouse gas emissions as calculated with the method.

The author says that he used the attributional LCA concept in this paper (in contrast to the consequential approach). In the scientific field of LCA, there are many arguments regarding the difference between these methodologies. The author should reflect these discussions more in the paper, but this does not significantly impact the importance and originality of the methodology presented in this paper.

4.3 Comments from Individual Reviewers

Individual reviewers of the External Review Panel recommend that the author shall consider the following points. These comments were not always supported by all Panel members.

1) The methodology discussion in the review process largely clarifies the issue of consequential/attributional LCA, but there are still some statements which are clearly an opinion of the author and therefore must be indicated as such. Reviewer 1 pointed out the following;

a) On page 4, last paragraph, the author states “Attributional LCA requires allocation of elementary flows between different product systems whenever there are product flows crossing the system boundaries of the investigated product system”. This is clearly the opinion of the author but not a methodological consensus, since many other scientists and practitioner would argue that also in attributional LCA allocation can be avoided by systems expansion, which is actually the preferred option in ISO 14040/44 (2006). It shall be stated that the statement is the opinion of the author and not a generally accepted view.

b) Page 5, second paragraph: Here the author mentions that the distinction between attributional and consequential LCA has emerged only recently. On the other hand he quotes a source from 2001 (Curran et al. 2001), which shows that this distinction is already quite established. 6 years (from 2001 to 2007) are a long time taken the overall history of LCA development into account. This is also confirmed by ISO 14040 (2006), Appendix 2, where it is clearly stated that both approaches have developed in recent years. Therefore the argument that the distinction has only emerged recently needs to be changed.

c) In Chapter 1.2. the author quotes several important references, but omits (Atherton, 2007), which is a key reference in this discussion, especially since it is a consensus document of the metals industry, including the aluminium and steel and iron industries. The arguments of this important paper have to be addressed in this methodology chapter as well (in addition to referencing it later in Chapter 2).

d) In Chapter 2.5.1 the discussion regarding consequential and attributional modeling is taken up again and on page 21, last paragraph, the author states "From a perspective of consequential system expansion, this is equivalent to the assumption that the use of metal from secondary production displaces secondary production outside of the product system instead of displacing primary metal production (Atherton 2007)". This may be the opinion of the author, but does not represent any form of scientific or industry consensus and it is not founded on the source given (Atherton 2007). To the contrary (Atherton 2007) shows that every amount of metal recycled displaces primary production, independent if the metal originally came from the primary or the secondary route. The statement needs to be changed so that it is clearly indicated as being the opinion of the author and that it is not based on (Atherton 2007).

2) In this paper, the methods to evaluate recycling systems were presented. It is important to point out that these approaches have several limitations. Although the author states those limitations in this paper, Reviewer 1 pointed out the following.

a) As stated e.g. in Chapter 2.5.1.1, directly after equation 12 , the author states that the recycled content or cut –off method only accounts for emissions that occurred during production and finishing of the material contained in the vehicle. In other words, this also means that the end-of-life phase of the automotive material is not considered, which demonstrates that the recycled content or cut-off approach does not cover the complete life cycle, since the implications of recycling are omitted. This needs to be addressed also in the methodology discussion in the report, apart from commenting on the formula.

b) In Chapter 2.5.1.1, last paragraph, the author states that the consideration of primary and secondary metal is based on the production route and not on the actual scrap content, for all alternatives considered in the model. On the other hand, in the preceding paragraph, where equation 13 is elaborated, he states that the avoided burden or system expansion method gives full emission credit for scrap and does not reflect the use of material from secondary production. In addition, the report in general and in several chapters stresses the importance of the methodological treatment of scrap for the overall result of any analysis. These points result in the fact that care has to be taken in selecting input data for comparative purposes (e.g. steel vs. aluminium) in regards to data symmetry, i.e. their suitability for being used in comparative analyses. This is important since the primary production route of aluminium does not use any scrap. In consequence any scrap use in the BF/BOF production route for steel has to be accounted for, e.g. by assigning a debit to the scrap use, otherwise the assessment would not be consistent and might lead to biased results. The author shall address this point and shortly explain the requirements for any data to be used (needs also to be mentioned in the related tables in Chapter 2.8., system boundaries). In this context, the terminology "cradle-to-slab" and "cradle-to-ingot" for the EAF slab steel product and the secondary aluminium ingot can be easily misunderstood. It is strongly suggested to change these terms to "scrap-to-slab" and "scrap-to-ingot" to avoid confusion.

Reviewer 2 pointed out on this issue also.

c) On page 23, “For $\alpha = 1$ equation (11) simplifies to equation (13). This is the avoided burden method from above, which gives full emission credit for prompt and end-of-life scrap recycling and does not reflect the use of secondary material.” The implications of this approach are significant for a comparative assessment. Use of $\alpha = 1$ can understate the burdens that are incurred and can favour one material over the other. The methodology properly allows for use of different α values but the user should justify their selection of the α value for a given application.

In addition for MSR it is unclear that this method would result in an overall balance in GHG emissions if it is applied for an automotive system and other product systems involved through material recycling. In other words it’s not clear whether GHG emissions will be conserved across systems.

5 Appendix B: Response to External Review Panel Report

The author of this report welcomes the results of the external review and all comments from individual reviewers. He would like to express his gratitude for the reviewers' expertise, scrutiny and constructive criticism, all of which added significant value to the report and the methodology it describes. As customary for a critical review according to ISO 14044 (2006), this chapter contains the author's response to the external review panel report.

5.1 Response to Results of the External Review

The author would like to express his full satisfaction and overall agreement with the results of the external review, reproduced in Section 4.2. He would like to assure the review panel that he is very much aware of the mentioned ongoing discussion in the field of LCA regarding the differences between and merits of attributional and consequential LCA methodology. However, he trusts that the relevant paragraphs in Sections 1.2 and 2.5 are sufficient for the purpose of this report. A manuscript with a detailed discussion of this issue is in preparation.

5.2 Response to Comments from Individual Reviewers

1) Responses to comments from Reviewer 1 regarding consequential/attributional LCA:

   a) The citation in this comment is incomplete. The complete statement is: "Attributional LCA requires allocation of elementary flows between different product systems whenever there are product flows crossing the system boundaries of the investigated product system. Clause 4.3.4.2 of ISO 14040/44 (2006) recommends avoiding allocation wherever possible through further division of unit processes or expansion of the product system." The sole aim of system expansion is to avoid that product flows cross the system boundaries, so that no allocation is necessary.

   b) Appendix 2 of ISO 14040 (2006) states: "Two possible different approaches to LCA have developed during the *recent* years. [Italics by author]" The author's statement "The rigorous and systematic methodological distinction between attributional and consequential LCA has emerged only recently“ thus merely reflects the view of ISO 14040 (2006).

c) Atherton (2007) is cited three times in Section 2.5.1, which contains a detailed explanation of the avoided burden approach (consequential system expansion) for the case of metals recycling. Section 1.2 only contains a generic treatment of the allocation issue in LCA, which is the reason why this publication is not mentioned there. Atherton (2007) is an industry declaration and does not contain a consensus of the scientific LCA community. Neither does it contain any proof or evidence for its statements.

d) This comment appears to be based on a misunderstanding of the last paragraph on page 22 (Section 2.5.1). The paragraph has been rewritten for clarification. It explains how the standard avoided burden approach for metals deals with *scrap inputs* to the product system, i.e. recycled content. Assigning a debit to secondary metal inputs that is equal to the difference between primary and secondary production emissions is equivalent to saying that using secondary metal does not reduce life cycle emissions. In the words of Atherton (2007): “If a designer specifies high recycled content in a well-meaning effort to reduce environmental impacts, it may stimulate the market to direct recycled feedstock towards designated products and away from production where recycling is most economical.” In rigorous consequential language this means that “the use of metal from secondary production displaces secondary production outside of the product system instead of displacing primary metal production”, which is the sentence that has been criticised in comment 1) d). In the author’s view this is a more rigorous way of saying “market stimulation is ineffective”, which again are the words of Atherton (2007).

2) Responses to comments regarding methods to evaluate recycling systems:

a) It is incorrect to say that the secondary content (cut-off) method does not cover the complete life cycle. It is important to distinguish between the emissions from the actual end-of-life management processes, i.e. collection, liberation, separation, etc., and the emission implications of generating scrap. Regardless of allocation/system expansion methodology, the former can and should be included if it is significant. As it happens, it is not significant in the case of passenger vehicle life cycles. It is also not correct to say that the implications of recycling are omitted in the secondary content (cut-off) method. The correct interpretation is that the secondary content (cut-off) method makes certain (strong) assumptions regarding the implications of recycling, just as the avoided burden approach.

b) The author fully agrees with the statement of Reviewer 1 in comment 2) b) that "[…] any scrap use in the BF/BOF production route for steel has to be accounted for." In fact, this comment inspired the author to extend the credit/debit system (CDS) (Section 2.5.1.1) to explicitly model the scrap inputs to primary metal production. As shown in equation (14), the old version of the CDS system is now a special case of the extended version. The author would like to thank Reviewer 1 for this excellent comment, which helped to further improve the methodology.

c) The author fully agrees with the comment of Reviewer 2 that "use of $\alpha = 1$ can understate the burdens that are incurred and can favour one material over the other." This is exactly the reason why the parameter α has been introduced in the methodology.
The author is of the opinion that the multi-step recycling (MSR) method results in an overall GHG emission balance if all *n* recycling cycles are considered.

Roland Geyer, Dipl. Phys., PhD — Santa Barbara, California, 7 December 2007