Modeling Material Inventories and Environmental Impacts of Electric Passenger Cars

Comparison of LCA results between electric and conventional vehicle scenarios

Master Thesis

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Abstract

This study presents a life cycle assessment (LCA) of electric passenger cars, which was carried out within the scope of a master thesis project. The focus of the LCA lies on the inventories of the electric drivetrain and the vehicle glider. Five main components of the electric drivetrain were analyzed in detail, based on supplier data. The storage battery was modeled with data from the Ecoinvent database. The materialization of the vehicle glider was assessed with data from the literature. The environmental consequences of lightweight construction concerning the glider were a further interest of the study. Hence, the glider materialization was modeled in a baseline and two different lightweight versions. The inventories of the main vehicle components were combined to model different electric vehicle scenarios. Each vehicle scenario was further modeled in an ICE diesel version, in order to make electric and conventional mobility options comparable.

The LCIA was calculated with the Swiss electricity mix, assessing the environmental impacts of each vehicle scenario with ten different damage indicators. It results that the contribution of the glider and the electric drivetrain to the total impacts of the electric car is high for many indicators. In the comparison of electric with conventional vehicles, the result depends on the applied indicator. The main conclusion is that electric vehicles cause higher human health impacts and damages to ecosystems, whereas conventional vehicles have higher impacts on global warming and the depletion of fossil resources. It was found that lightweight construction could be effective in reducing environmental impacts of passenger cars. However, the effect depends on the choice of lightweight materials for the substitution of steel and iron.

The product system was modeled with two different allocation approaches for end-of-life processes – a cut-off and a system expansion and substitution model –, which are compared by means of a sensitivity analysis. It is concluded that the modeling approach has a high influence on the LCA results. In the substitution model, recycling processes substituting primary materials improve the environmental performance of the electric vehicle compared to the cut-off model. Further, a sensitivity analysis concerning the electricity supply mix for the use phase of the electric car was calculated. It can be concluded that electric mobility is only a sustainable strategy for climate change mitigation if the electricity used to power the vehicle is produced from renewable sources.

List of Abbreviations

BEV	Battery Electric Vehicle
BFE	Swiss Federal Office of Energy
BFS	Swiss Federal Statistical Office
CED	Cumulative Energy Demand
CExD	Cumulative Exergy Demand
CNT	Carbon Nano Tube
CO_2-eq	CO_2 equivalents
DALY	Disability Adjusted Life Years Lost
EI 99	Ecoindicator 99
EI-pt	Ecoindicator points
e-mobility	Electric mobility
e-motor	Electric motor
EOL	End Of Life
EOLV	End Of Life Vehicle
EV	Electric Vehicle
GHG	Green House Gas
GWP	Global Warming Potential
HEV	Hybrid Electric Vehicle
ICE	Internal Combustion Engine
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MJ-eq	Mega Joules equivalents
NEDC	New European Driving Cycle
PAF	Potentially Affected Fraction
PDF	Potentially Disappeared Fraction
PDU	Power Distribution Unit
PHEV	Plug-in Hybrid Electric Vehicle
PM_{10}	Particulate Matter $\leq 10\ \mu m$
PNGV	Partnership for a New Generation of Vehicles
THELMA	Technology-centered Electric Mobility Assessment
UCTE	Union for the Coordination of Transmission of Electricity
vkm	Vehicle kilometer

1 INTRODUCTION

Electric mobility can be seen as a potential strategy to solve the environmental problems caused by road transportation, as electric vehicles have quiet engines, are tailpipe emission free and allow for the use of different (renewable) energy sources (Helms, Pehnt et al. 2010). Electric cars for individual mobility have become a prominent issue in the light of climate change and the debate on the availability of oil resources. There is widespread consensus on the need of increasing the share of renewable energy sources in the transportation system in order to reduce environmental impacts and the dependency on (foreign) fossil resources. However, the question remains whether and how electric cars can potentially contribute to solve these problems.

In Switzerland, the overall growth of mobility in the last ten years has been dominated by the increase of personal mobility in terms of kilometers travelled per person and year. On average, Swiss citizens travel 37 km per day (during one hour and 40 minutes), of which 67% are travelled by car, 16% by train and 5.5% by walking. Interestingly, almost half of the distance travelled per day is for the reason of leisure activities, followed by travelling for work and education with 27% and shopping with 11% (BFS 2010b). Most of the distances travelled by car are actually rather short: 45% are not longer than 5 km and only one third are longer than 10 km (BFS 2007). Similar conclusions can be drawn from a survey by TÜV Rheinland on the acceptance of electric mobility in Germany, finding that 61% of respondents do not travel more than 50 km per day, 30% between 50 and 100 km and only 9% drive longer distances per day (Forum ElektroMobilität e.V. 2010). These results show that for most people the current range of electric cars' batteries would be sufficient for average travel distances. In 2005, 81% of Swiss households owned their own car and in 2009 4 Mio passenger cars were registered in Switzerland (in view of the population size of 7.8 Mio people (BFS 2010a)) (BFS 2010b). In the year 2000, the Swiss Federal Statistical Office (BFS) counted 3.7 Mio people in the active population and nearly half of them commute to work and back by car (BFS 2010c). On average, travels to work are about 12 km long (BFS 2007). In terms of energy, in 2009 35% of total energy has been used in the transport sector, compared to 29% for households, 19% for industry and 16% for the service sector (BFE 2009). The energy used for mobility (including air traffic) is covered almost entirely by fossil sources (96%). Respectively, electricity used for rail traffic covers only 4% of total transport energy (BFS 2010b). CO_2 emissions due to the Swiss transport system sum up to about 17 Mio tons per year of which the largest share is caused by passenger cars (67%) (BFS 2010b). PM_{10} emissions of passenger cars have decreased from 2000 t in 1980 to about half a ton per year in 2005. Accidents, health impacts, climate impacts and noise are the most important reasons for external costs caused by road traffic (BFS 2010b).

Summarizing, the Swiss traffic system is heavily dependent on fossil fuels and dominated to a large extent by passenger cars. The numbers show that average distances travelled per day are rather short and covered to two-third by car. Internal combustion engine (ICE) cars cause various external costs because of health impacts, emissions of greenhouse gases (GHG) or noise exposure (BFS 2010b). Hence, electric cars seem to have the potential to overcome these problems of personal mobility. They are quiet, tailpipe emission free, and – if powered by electricity from renewable sources – use phase GHG emissions are much lower. Although their range is limited by the storage capacity of the batteries, they could actually cover most everyday trips in Switzerland.

Nevertheless, it remains questionable whether the environmental performance of electric cars really proves to be better compared to conventional cars if impacts from the production, use and disposal of the vehicle are analyzed in detail. For this purpose, life cycle assessment (LCA) is a powerful tool. Hence, it is the goal of this master thesis to establish a precise model of the material composition of electric passenger cars in order to assess impacts from the production, the use phase and the disposal of an electric vehicle (EV). A first screening of the literature on environmental impacts of e-mobility lead to the hypothesis that so far, most studies have focused on the comparison of use phase impacts (mainly GHG emissions) between electric and conventional cars (Boureima, M. et al. 2009; Samaras and Meisterling 2008). Further, environmental impacts from the production of the storage battery have been assessed (Notter, Gauch et al. 2010; Zackrisson, Avellán et al. 2010). A more in depth literature review (see chapter 4) could verify this assumption, leading to the conclusion that to date, comparisons between electric and conventional cars have analyzed primarily impacts from the use phase and the battery production. Therefore, this project specifically focuses on the material composition of the electric propulsion system and the vehicle infrastructure. Environmental impacts of electric and conventional ICE cars will be compared from a life cycle perspective for various impact categories. In line with this focus, only battery electric vehicles (BEV) will be considered, but not hybrid or plug-in hybrid electric vehicles (HEV/PHEV).

2 METHOD

The method applied to answer the research questions posed in chapter 3 is Life Cycle Assessment (LCA), based on the ISO 14040/44 standard. As a methodological guideline for this study serves the 'Hitch Hiker's Guide to LCA' (Baumann and Tillman 2004), as well as the specifications in the ISO standards (ISO 2006a; ISO 2006b).

This chapter gives an introduction to the method and describes the theoretical concept of LCA in more detail in the following paragraphs. This description of the method is mainly based on the explanations in the 'Hitch Hiker's Guide to LCA' (Baumann and Tillman 2004). Hence, if not otherwise specified the publication by Baumann and Tillman (2004) serves as the general reference for the entire chapters 2.1 and 2.2.

2.1 Basic concept of LCA

The basic idea of life cycle assessment, which is sometimes also called cradle-to-grave analysis, is to reveal all environmental impacts a certain product causes throughout its whole lifetime. Hence, the life cycle describes all processes from raw material extraction to produce the product to its final disposal. When all in- and outputs to and from a product system are assessed, the impacts to the environment caused by this product can be calculated. LCA usually analyzes a variety of different types of environmental impacts, such as global warming, resource depletion, or damages to ecosystems, and also impacts to human health (for example due to toxic substances) are considered as environmental impacts.

LCA has been recognized as a useful tool to analyze product systems from an environmental perspective. It can be applied to a wide variety of different situations and with many different intentions. The method can for example be helpful to learn more about a certain product system or it can support the decision making process when different types of a product are compared. Depending on the intended application of the study, LCA can vary from very short screening-type studies providing a rough overview to in-depth long-term studies analyzing a product and its impacts in detail.

2.2 LCA in detail

The procedure of carrying out an LCA is organized in the following four main steps: goal and scope definition, inventory analysis, impact assessment and interpretation.

2.2.1 Goal and scope definition

At the beginning of any LCA study the goal of the study has to be defined precisely, i.e. the research question that should be answered by carrying out the LCA needs to be formulated. This has been reported in chapter 3 for this project. Afterwards, the scope has to be defined, which describes how the study should be carried out and how the product system of interest should be modeled (to compare scope definition see paragraph 3.2). This includes the definition of the following aspects:

The functional unit: In order to make all the inputs to, or emissions from the investigated product comparable, they have to be put in relation to a specific function of that product. In the case of this project, the functional unit is defined as driving an electric passenger car for one kilometer on Swiss roads. All in- and outputs will then be measured as the amount per functional unit.

System boundaries: It has to be defined which processes belong to the product system. The system has also to be delimited in terms of its geographical as well as temporal boundaries. It has to be specified for what geographical region and for which temporal horizon the study should be valid.

Allocation: The allocation approach that will be applied to model the product system has to be chosen. In this study, the inventory of the product system will be modeled with two different allocation methods, which are the cut-off approach and system expansion and substitution. The influence of the allocation approach on the results will then be tested by means of a sensitivity analysis.

Initial flowchart: As soon as the functional unit and the system boundaries have been set, an initial product flowchart can be drawn. This helps to structure the procedure of the study and to depict what is already known about the product system. In order to increase the level of detail, the system can further be subdivided into a foreground and a background system (see **Figure 1**).

Data quality: Depending on the product and the purpose of the LCA study, the required quality of the data needed may vary. For the one study, average data from the literature might be accurate enough, whereas for another study site-specific data from manufacturers might be required.

2.2.2 Inventory analysis

The life cycle inventory (LCI) is basically a balance of all mass and energy flows into and out of the system. All these flows are listed and calculated in relation to the functional unit. To do so, data is collected for all relevant activities throughout the product life cycle, such as production, transportation, use and waste treatment processes. For each process, the required materials and energy carriers as well as the arising emissions and solid wastes have to be assessed. The decision on which processes are of importance depends on the specifications in the goal and scope definition, for example the selection of system boundaries or data quality requirements. The inventory can be supported by a more detailed system flowchart, depicting the relevant processes within the system and the mass and energy flows between them.

The inventory analysis may be complicated by the need to *allocate* certain flows between different products. For example, if the product of interest is transported together with another product: will then all transport emissions be allocated to the one product of interest, or should they rather be split between the two products? And according to which *allocation rule* should they be split? Allocation also becomes necessary when waste materials are recycled and reused instead of the primary material. Since this is quite a complex issue and different allocation rules are sometimes controversial, it would be out of scope to provide a comprehensive description thereof. The allocation approach applied in this study will be described in more detail in paragraph 5.3 from a practical point of view. Further information on allocation can be found in chapter 4.4 in the 'Hitch Hiker's Guide to LCA' (Baumann and Tillman 2004).

2.2.3 Impact assessment

The result of the LCI is an extensive list of substances – amounts of resources extracted from and pollutants released to the environment. This list is generated by cumulating the inventoried amounts of material and energy flows and relating them to resource inputs and emission outputs. However, the inventory results per se do not yet tell anything about the real impact caused to the environment. Therefore, the LCI has to be translated and aggregated into more meaningful indicators of environmental impacts in the life cycle impact assessment (LCIA). This is done in the series of steps stated below, some of which are compulsory whereas others are optional according to the ISO standard.

Classification: In the first step of the impact assessment the inventory results are classified according to the type of environmental impact they cause. Generally, the following three main areas of impacts are considered in most impact assessments: resource use, human health and ecological consequences. However, the set of more specific impact categories applied can vary depending on the type of impact assessment method that is used. One of the most prominent methods in Europe is *ecoindicator 99* (EI 99), which will be applied in this study. It has to be mentioned though, that recently the new impact assessment method *ReCiPe* has been developed, which can be seen as the subsequent method of ecoindicator 99. ReCiPe[1] combines the midpoint and the endpoint approaches of the CML and ecoindicator methods. In this study however, ecoindicator 99 will still be applied instead of ReCiPe in order to make the impact assessment results comparable to earlier studies on the same topic. The impact assessment with ecoindicator evaluates environmental impacts in the categories listed in **Table 1**.

Table 1 Main impact categories and subcategories (with units) for impact assessment with ecoindicator 99

Human health	Ecosystem quality	Resources
Carcinogens (DALY)	Ecotoxicity (PAF*m^2yr)	Minerals (MJ surplus)
Respiratory organics (DALY)	Acidification/ Eutrophication (PDF*m^2yr)	Fossil fuels (MJ surplus)
Respiratory inorganics (DALY)	Land use (PDF*m^2yr)	
Climate change (DALY)		
Radiation (DALY)		
Ozone layer (DALY)		

(DALY=Disability Adjusted Life Years Lost, PAF=Potentially Affected Fraction, PDF=Potentially Disappeared Fraction)

Applying ecoindicator, every substance of the LCI is assigned to the specific categories of environmental impacts it causes, according to the categories listed in **Table 1**. In some cases, it might also make sense to analyze only one single type of environmental impact, e.g. if the study has a specific focus on one environmental issue. An example of such a single indicator that is very often used is *global warming potential* (GWP), measured in *kilograms of CO_2-equivalents* (kg CO_2-eq). And often, it is required to apply several indicators in order to get the whole picture of a product's environmental performance.

Characterization: After classifying the impacts, they have to be characterized quantitatively in the next step. This means that all substances contributing to the same impact category have to be translated from a mass or energy load into an impact load, ending up with one specific unit for each category (as in **Table 1**). This makes it possible to sum up all impacts per category into one single indicator for this category. This translation happens using so-called *equivalency factors* (also called *characterization factors*), which are based on models of cause-effect-chains that describe the behavior of a substance in the environment. Taking the impact category GWP as an example, all GHG will be put in relation to the effect of CO_2, leading to one single indicator for global warming with the unit kilogram CO_2 equivalents.

[1] For more information on ReCiPe see: http://www.lcia-recipe.net/

Normalization (optional): The impacts per category can be normalized to a certain magnitude of impact, for example to the total impacts arising in a country. This step can help to put the assessment results into a meaningful relation.

Grouping (optional): After characterization, it is usually possible to summarize the specific impact categories into the three main categories *human health*, *ecosystem quality* and *resources* (depending on the characterization method other categories are possible as well). Another possibility to structure the results of the impact assessment is to group them for example as global/regional/local impacts. This might make the communication of results more intuitive.

Weighting (optional): In a next step, the environmental effects of the chosen impact categories can be weighted against each other, depending on the relative importance attributed to each impact. Different *weighting factors* can be applied and the underlying principles of how these factors are derived usually stem from the social sciences. According to the ecoindicator approach, there are three different perspectives to define weighting factors, which are applied as well in the classification and characterization steps: the individualist, the hierarchist and the egalitarian perspective, a concept derived from cultural value theory. Weighting makes it possible to aggregate all impacts that have been calculated in the LCIA into one single number. Hence, the results of the LCA study can be presented as one single score of environmental impacts that are caused by the product that has been analyzed. In the case of ecoindicator, this single score is expressed in ecoindicator points (EI-pt).

2.2.4 Interpretation

After establishing the inventory and assessing the environmental impacts based on the inventory, the results have to be further analyzed, presented and discussed. In this last phase of the LCA study, it is important to validate the quality of the results. A sensitivity analysis can be performed in order to check the robustness of the results with respect to certain parameters and choices. Similarly, it has to be analyzed whether the input data for the impact assessment is accurate enough to draw certain conclusions. If for example, it results that a certain manufacturing process of the product causes significant impacts on human health, it has to be verified if the data used to model this process is good enough to allow such a conclusion. Finally, the LCA results have to be presented graphically and discussed with respect to methodological choices, conclusions and consequences.

2.3 Databases and software

It becomes clear that it can be a very difficult and time-consuming task to establish the inventory and calculate the impacts by simple spreadsheet software, especially the more complex a product system is. In practice, it would be almost impossible to collect all the data needed to model every underlying process in the manufacturing chain of a product. Therefore, LCA is usually performed with the help of software tools and existing databases. There is a variety of different LCA software and databases available and concerning databases they are usually designed for a specific geographical region. For this master thesis project, the LCIA will be calculated with the SimaPro software (PRé Consultants 2010) and with data from the Ecoinvent database (current version v2.2) (Swiss Centre for Life Cycle Inventories 2010).

The Dutch company *PRé Consultants* has developed the SimaPro software (as well as the Ecoindicator impact assessment method). SimaPro allows the user to either implement data from a database of choice (e.g. Ecoinvent), or to create his own datasets with manually collected data. The software supports the whole LCA procedure, from establishing the LCI to calculating the LCIA. In the impact assessment, every step can be calculated separately with various different impact assessment methods. Results are displayed either graphically or as a table and can be exported to Excel for further editing.

The Ecoinvent database supplies international LCI data on energy supply, resource extraction, material supply, chemicals, metals, agriculture, waste management services, and transport services. It is used in more than 40 countries worldwide and is compatible with all major LCA and eco-design software tools (Swiss Centre for Life Cycle Inventories 2010).

3 CASE STUDY: LCA OF ELECTRIC MOBILITY

The main part of this master thesis consists of applying the methodological concept of LCA as presented in chapter 2 to a case study of electric mobility. The general goal is to assess the environmental impacts of battery electric vehicles (BEV) with a focus on glider and electric drivetrain materialization and to compare the impacts with those of conventional cars.

3.1 Goal of the project

According to the findings in the literature review (see chapter 4), most previous LCA studies of (electric) cars are based on a vehicle model derived from LCI data of the Volkswagen Golf A4 (Schweimer and Levin 2000). Therefore, this vehicle model will be compared with data on vehicles' material composition from other studies to come up with different scenarios of the glider's materialization. Further, it is crucial to establish an inventory of the electric drivetrain, based on supplier data. It is the goal of this master thesis project to calculate life cycle environmental impacts of BEV based on the collected LCI data for different vehicle scenarios and to compare LCIA results between electric and ICE vehicle scenarios.

Main research question

What are life cycle environmental impacts of electric passenger cars, and what are the contributions of the glider and the electric drivetrain to the overall impacts?

This main research question is completed by a set of subquestions, which will have to be answered in order to achieve a conclusion on the posed research question.

Subquestions

- What is the material composition of the different components of the electric drivetrain?
- What is the material composition of the glider of the average passenger car?
- What are possible scenarios of the glider's material composition in terms of lightweight construction?
- What are implications for the energy consumption in the use phase, depending on the modeled material composition and respective vehicle weight of EV?
- What are implications for end-of-life processes of EV depending on the material composition?
- What is the difference in life cycle environmental impacts between electric and ICE passenger cars?

3.2 Scope of the life cycle assessment

The intended application of the LCA study of this project is to be part of an overall technology assessment of electric mobility and to contribute to the assessment of the feasibility of this technology. Hence, the target group in mind is the interested public and scientific audience.

The study is conducted with a specific focus on Switzerland. Accordingly, the use phase of the vehicles as well as parts of the production and end-of-life (EOL) phases are modeled according to the

Swiss situation. However, the geographical system boundary is defined as beyond Swiss national borders, i.e. international. The assessment should be valid for the present time.

The products to be analyzed with the LCA method are primarily different classes of passenger electric vehicles. The classes of EV should differentiate between vehicle size and materialization. Accordingly, different vehicle scenarios will be compared against each other. Hybrid and plug-in hybrid EV are not part of the scope of analysis and will not be considered. The different classes of EV modeled should represent average vehicles of the classes of mini cars, city cars and compact (or golf-class) cars. Each electric car of a certain class modeled should finally be compared to an ICE car of the same size, i.e. the compared vehicles should only be different in terms of their powertrain and energy consumption. The Swiss power supply mix is applied as energy source to charge the electric car in the use phase, since the study is conducted from a Swiss perspective and the main focus lies on vehicles' materialization. The influence of differences in the electricity supply mix on the LCIA results will be considered with a sensitivity analysis, however. The type of battery considered for the EV is a lithium-ion battery of the $LiMn_2O_3$ type, Ni- and Co-free, as in Notter, Gauch et al. (2010). With respect to the use phase of ICE cars, only one type of fossil fuel is considered, which is diesel. The reason for this constraint is to focus the discussion on vehicle characteristics and not to discuss impacts of different fossil or bio-fuels.

As a starting point of analysis and data collection, a preliminary product system flowchart was drawn (see **Figure 1**), indicating the functional unit of driving an electric passenger car for one kilometer on Swiss roads. Hence, the corresponding reference flow is defined as 1 vehicle kilometer (vkm). The system is subdivided into a foreground and a background system. The foreground system includes the three main life cycle phases of production, use and EOL of the product and represents all processes under direct influence by measures and choices taken within these three phases. In the background system, all inputs and outputs to and from the foreground system are modeled, representing all processes directly influenced by choices concerning the foreground system.

With respect to data quality, data needs to be more comprehensive for processes of the foreground system. The production of the electric drivetrain is analyzed by collecting detailed data on resource and energy use from suppliers. The production of the glider is modeled with average data from the literature. Impacts arising from the use phase can be assessed by modeling the energy consumption of the various cars in the *new European driving cycle* (NEDC) based on grid-to-wheel efficiency, car mass and geometry. Further, the energy consumption for non-propulsion purposes (e.g. heating, cooling, light, radio,…) is also included. Finally, EOL processes for all vehicle components are analyzed with respect to recycling and waste disposal. Processes in the background system are modeled with data from the Ecoinvent database v2.2. Having collected all data for the inventory, life cycle impacts of EV can be calculated with the Simapro software.

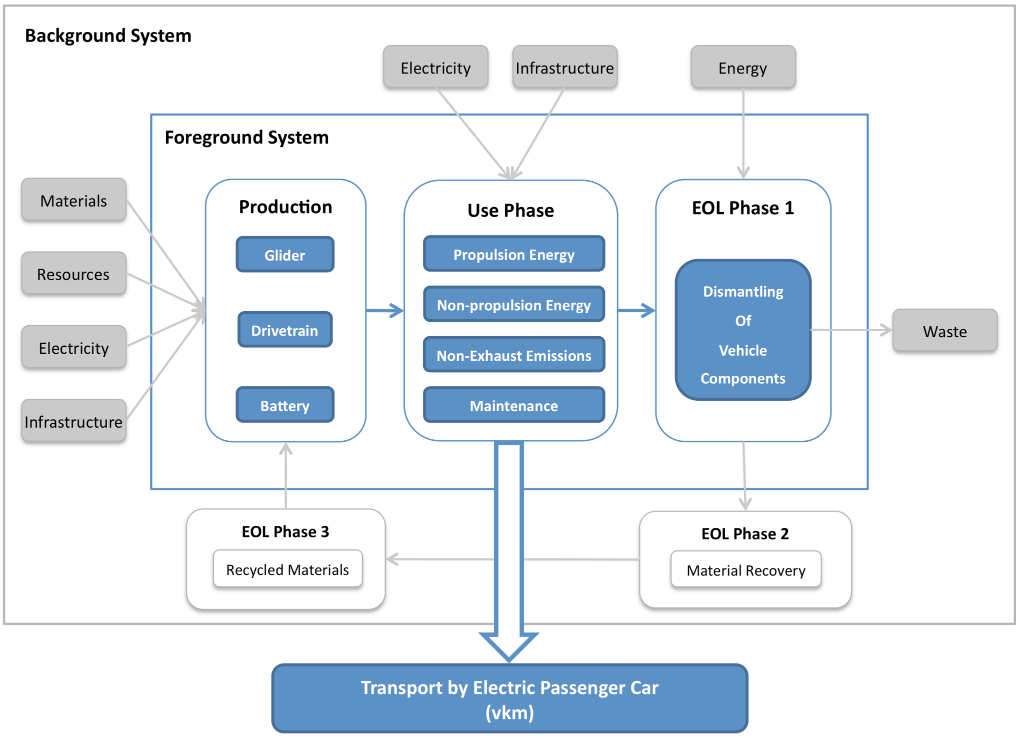

Figure 1 Preliminary product system flowchart

Background and foreground system, functional unit of driving an electric passenger car on Swiss roads (1vkm)

3.3 Overarching research project

This master thesis will be conducted within the context of the THELMA research project[2]. THELMA stands for *Technology-centered Electric-Mobility-Assessment* and is a cooperation of research institutes at EMPA, PSI and ETH. The main goal of THELMA is to assess the life cycle based environmental performance of electric vehicle technologies in comparison to several combustion engine options (fossil, biofuels, hydrogen) in terms of environmental impacts.

This master thesis will be part of work package 1 (WP1) of the THELMA project at the EMPA Life Cycle Assessment and Modeling Group. The main task of WP1 is to assess the LCA-based environmental performance of vehicles and energy supply chains. Research shall focus on technologies and materials related to e-mobility, allowing for a comparison of competing vehicle technologies. Within this framework, goals of WP1 are to model environmental inventories of electric vehicles from a life cycle perspective, to assess life cycle impacts per kilometer of transport and to compare LCA results of electric cars with conventional reference technologies and other new mobility technologies.

[2] For more information on THELMA see: www.thelma-emobility.net

4 LITERATURE REVIEW

This chapter intends to give an overview on recent findings of research on electric mobility. Results from the areas of life cycle assessment of (electric) passenger cars, vehicle design and materialization, battery production as well as end-of-life processes are summarized. However, this review of the literature is not meant to be complete, but should give an overview of some interesting issues and findings of electric mobility research.

4.1.1 Life cycle impacts of electric vehicles

A large number of studies considering life cycle environmental impacts of electric vehicles as alternative mobility solution have already been published. However, research on environmental impacts of electric mobility so far has focused mainly on the use phase of the vehicles and the production of the batteries. Accordingly, comparisons between EV and conventional cars are dominated by impacts depending on the respective power sources for propulsion. Concerning the vehicle body and chassis, many studies are based on an analysis of the Volkswagen Golf A4 (Schweimer and Levin 2000) (see 4.1.2) due to the lack of more up-to-date data on vehicle materialization. So far, it has been shown that e-mobility can be beneficial compared to fuel-based mobility, given that renewable energy sources for electricity production are used (Notter, Gauch et al. 2010).

In a comparative study on life cycle emissions of electric and ICE vehicles for Germany by Helms et al., the authors apply the model of the VW Golf 4 for the assessment of the vehicle production for both electric and ICE vehicle (Helms, Pehnt et al. 2010). They find that the highest contribution to life cycle GHG emissions stems from the use phase for both types of cars. Nevertheless, vehicle production is also of relevance for GHG emissions, especially for EV because of their battery. In an assessment of life-cycle impacts of Li-Ion batteries, Notter et al. have shown that the battery's contribution to total environmental impacts of EV is about 15% (Notter, Gauch et al. 2010). Life cycle GHG emissions of a BEV (around 22'000 kg CO_2-eq) calculated with the German average electricity mix are slightly higher than emissions of an ICE diesel car and slightly lower than those of an ICE petrol car (Helms, Pehnt et al. 2010). However, if electricity produced by a coal plant is used for charging the BEV, total GHG emissions rise up to 40'000 kg CO_2-eq. Only by using renewable energy sources to charge the battery, GHG emissions can be reduced significantly, i.e. down to 7000 kg CO_2-eq if wind is used as power source. In that case, GHG emissions arise almost solely from the vehicle production (Helms, Pehnt et al. 2010). In a similar study with a Swiss perspective, Althaus and Gauch compare life cycle impacts of BEV and ICE vehicles, as well based on the model of a VW Golf 4 (Althaus and Gauch 2010). In their assessment of life cycle GHG emissions, the emissions of the BEV (0.09 kg CO_2-eq/km) are substantially lower than those of a diesel or petrol ICE vehicle (0.2/0.25 kg CO_2-eq/km), if the EV is charged with the Swiss electricity mix. However, global warming potential (CO_2-eq/km) as well as impacts on human health (EI99 DALY/km) and ecosystem damages (EI99 PDF*m^2yr/km) of the EV rise to the same or even a higher level compared to the ICE car if the car is charged with electricity from the UCTE-mix or a coal plant. Total life cycle impacts are assessed with ecoindicator 99 for the categories of human health damage, loss of ecosystem quality and resource depletion. Calculations with the Swiss power mix result in slightly lower impacts of BEV concerning the damage to human health, and similar impacts concerning ecosystem damage, compared to impacts of diesel or petrol ICE cars. However, considering resource depletion, the difference in impacts is substantial, with BEV having three to four times lower impacts than fossil ICE vehicles.

Boureima et al. have compared life cycle impacts of BEV, HEV, liquefied petroleum gas and gasoline cars representing the Belgian family car fleet (Boureima, M. et al. 2009). They find that total GHG emissions of BEV are 27.5% lower compared to those of gasoline cars and that BEV also have the lowest impact on human health and air acidification. For all types of cars, manufacturing processes were assumed to be the same except for the difference in powertrain technologies. The use phase has the highest impact on life cycle GHG emissions for all vehicle types. Concerning BEV, impacts on human health depend heavily on the type of electricity mix applied in the use phase. By replacing the Belgian mix by a renewable mix (half wind, half hydropower), impacts on human health could be lowered more than five times.

Concluding on these findings, the battery only contributes to a low share to overall environmental impacts of EV. Impacts originating from the use phase of the vehicle depend substantially on the power source for electricity. Hence, it can be concluded that the electric propulsion system and the body and chassis of electric cars are significant contributors to overall environmental burdens of electric vehicles if clean power sources are used for the use phase.

4.1.2 Life cycle impacts of conventional cars

Studies investigating life cycle impacts of conventional ICE passenger cars are of interest for this project mainly because of information on the material contents of all non-propulsion components of a vehicle, as well as for the comparability of LCA results.

One of the most fundamental and detailed life cycle assessments of a passenger car is the LCA of a Volkswagen Golf A4 by Schweimer and Levin (2000), which has served as a basis for many studies and has often been referred to. The authors have found that approximately 9% of the total energy is needed to manufacture the vehicle, 12% to mine the materials and 8% to provide the required fuel. The remaining 71% are attributed to the use phase of the car's life cycle. Concerning atmospheric emissions, CO_2, CO and NO_X are dominating in the use phase, while the majority of hydrocarbon and sulfur oxide emissions occur in the production phase and the distribution of fuel. Metals and chlorine are emitted predominantly during the mining of raw materials. It has to be mentioned though that the authors declare some data in the LCI to be outdated or uncertain (Schweimer and Levin 2000).

Castro et al. have analyzed life cycle impacts of the average passenger vehicle (with average weight and average material composition) of the Netherlands, focusing on the current dismantling and recycling practice in the country. It results that the largest environmental impact (measured with EI 99) of the vehicle's life cycle occurs in the use phase with over 90% due to the combustion and depletion of fossil fuels (Castro, Remmerswaal et al. 2003). Spielmann and Althaus have analyzed environmental impacts of conventional ICE cars in an LCA of Swiss passenger cars (Spielmann and Althaus 2007). Referring to Spielmann et al. 2004, the authors state that for the Swiss average passenger car representing the year 2000 fleet, transport emissions of the use phase, such as CO_2 and NO_X, dominate life-cycle impacts with a contribution to total impacts of 85% and 75% respectively. However, the car infrastructure contributes considerably to the overall score (10% to total CO_2, 20% to total NO_X). Therefore, the overall impact of car transportation is sensitive to changes of specific material and fuel expenditures in car manufacturing (Spielmann and Althaus 2007). Looking at options to improve the environmental performance of passenger cars, lightweight construction could be one among others, reducing the fuel consumption in the use phase (Leduc, Mongelli et al. 2010). Because of the fact that greater amounts of aluminum are used in lightweight vehicles, the authors expect an increase of the amount of waste and PM emissions associated with the production phase. A weight reduction of 30% would lead to a decrease in environmental impacts from 5% to 15% (depending on the impact category) compared to the baseline scenario. For the impact categories of PM emissions and solid waste however, impacts would increase by 2% and 8% respectively (Leduc, Mongelli et al. 2010).

4.1.3 Vehicle design, components and materials

Concerning weights and materialization of ICE passenger cars, Spielmann and Althaus have summarized results from various LCA studies (Spielmann and Althaus 2007). The average weight of vehicles sold in Switzerland has grown from 1300 kg in 1995 to 1500 kg in 2005. As a trend extrapolation until 2010, the authors assume the average vehicle weight to stabilize on the 2005 level, arguing that the trend towards lightweight materials outbalances the trend to bigger vehicles with more safety and comfort features. The material composition of the average passenger car, based on the LCA study of the Volkswagen Golf A4 (see 4.1.2), is composed of 73.7% ferrous compounds, 17.4% plastics and textiles, 3.9% aluminum and 2.4% non-ferrous metals. Life-cycle impacts (measured with EI 99) of the car infrastructure are dominated by heavy metal emissions of cadmium and arsenic in lead production and by the depletion of crude oil and natural gas. However, the authors identify a lack of knowledge concerning the development of car materialization.

Castro et al. report a tendency towards an increased application of polymers, aluminum, magnesium and composite materials as well as ultra-strong steel alloys in vehicle construction, in order to reduce the vehicle weight (Castro, Remmerswaal et al. 2003). The authors state that lightweight metals are well recyclable; other lightweight materials however could pose a challenge to the recycling industry. Further, it is assumed that many lightweight materials such as aluminum, magnesium or carbon-fiber are more energy intensive and cause higher GHG emissions in the production process than conventional steel (Koffler and Rohde-Brandenburger 2010).

A comparison of the material contents between one baseline and two lightweight vehicle scenarios can be found in Schmidt, Dahlqvist et al. (2004). The reference vehicle represents a 1000 kg midsized car with the material range of EOL vehicles produced in the early 1990's. The material compositions of the two lightweight cars of 900 kg and 750 kg weight are indicated as ranges instead of absolute values, combining various lightweight strategies. Material contents are given as a share of the total vehicle weight. The reference vehicle contains 70% ferrous metals, 3% aluminum and 15% plastics and textiles. For the 900 kg car scenario, ferrous metals range from 34% to 60%, aluminum from 11% to 33% and plastics and textiles from 13% to 23%. The lightweight car of 750 kg consists of 15%-49% ferrous metals, 10%-40% aluminum and 23%-32% plastics and textiles (Schmidt, Dahlqvist et al. 2004). A similar study comparing one baseline (1994) and three lightweight prototype cars has been conducted by Tonn, Schexnayder et al. (2003). The study focuses on waste issues associated with the adoption of new lightweight vehicles. Ford, Daimler Chrysler, and General Motors developed the investigated prototypes with the goal of a 40% reduction of the vehicle weight. From the inventory of the material composition of these prototype cars it can be seen that the amount of aluminum increases substantially, while the amount of ferrous metals is much lower, compared to the baseline car. The amounts of magnesium, plastics and carbon fiber vary a lot depending on the lightweight scenario. The assessment shows that the total waste stream of each of the lightweight cars is 60%-80% greater than that of the baseline car. Generally, the authors find plastics to be the major contributor to regulated hazardous wastes (Tonn, Schexnayder et al. 2003).

In recent years, the topic of availability and environmental impacts of rare earth metals used in various (new) technologies has become an issue. Concerning passenger cars, rare earth metals are important raw materials for the production of several components. Wäger and Lang state that around 50% of the primary production of platinum group metals (such as platinum, palladium and rhodium) is used in vehicle catalyzers (Wäger and Lang 2010). Rare earths, especially neodymium, samarium, gadolinium, dysprosium and praseodymium, are part of permanent magnets applied in electric motors. The mining of rare earths from the mineral monazite can be extremely hazardous because of the release of radioactive elements. Lithium is at the moment the most promising element for the chemistry-basis of second-generation batteries, i.e. for the application in EV, because of its particular physical characteristics (see 4.1.4). The extraction of lithium takes place either from brine or from lithium-containing minerals. Due to the expected increase in lithium deployment in new technologies, an expansion of mining activities to so far untouched landscapes will probably be necessary, with the consequential ecological effects (Wäger and Lang 2010).

4.1.4 Battery production

Besides impacts arising from the use phase, the production of the battery is crucial in a comparison of EV with ICE cars. Today's most promising batteries are almost entirely based on lithium chemistries due to the high energy storage capacity and low weight. The 2010 U.S. geological survey estimates that globally, 23% of the lithium in end-use markets is used for batteries (U.S. Geological Survey 2010). According to Notter et al., the battery contributes 15% to total environmental impacts of EV, measured with ecoindicator 99 (Notter, Gauch et al. 2010). Based on a detailed LCI of a Li-Ion battery, they find that impacts caused by the lithium extraction are less than 2.3% of the total impacts. The highest impacts caused by the battery stem from the supply of copper and aluminum for the production of the anode and the cathode. However, the authors' analysis reflects a specific type of production process, which is optimized in terms of energy-efficiency. Since this process is not yet standard for battery production in general, environmental impacts due to battery production could be considerably higher in other cases because of a higher energy consumption. Based on their study, the authors conclude that EV can be environmentally beneficial compared to conventional cars and that the Li-Ion battery does not lead to an overcompensation of the potential benefits of BEV.

Zackrisson, Avellán et al. (2010) have calculated life cycle impacts of a fictitious 10 kWh lithium iron phosphate ($LiFePO_4$) battery designed for PHEV application. They conclude that in the production phase, global warming impacts are dominated by energy use in manufacturing (>50%), electronics (30%) and the cathode (10%). Transportation of raw materials and components has little (3%) impact. Further, they compare the use of water with N-methyl-2-pyrrolidone (NMP) as a solvent in the slurry for casting cathodes and anodes of lithium-ion batteries and find that it is environmentally preferable to use water.

In battery production, the most crucial design elements for the energy storage system include battery life, thermal design, crash worthiness and the design and architecture of the monitoring system (Guerin and Leutheuser 2010). Improvement of the storage batteries in terms of provided range is still a high priority in battery technology research. Han et al. have developed a 28 Ah plastic lithium-ion cell for EV applications with the goal of providing a competitive driving range to that of ICE cars. They could demonstrate a high specific energy (160 Wh/kg), high specific power (526 W/kg at 80% DOD), excellent round-trip energy efficiency (92%), and low self-discharge rate (6% in 30 days), which are considered as promising results (Han, Seo et al. 2001). In the development of high-capacity lithium-ion batteries the application of carbon nanotubes (CNT) has become an option. According to Robertson (2004) in Köhler, Som et al. (2008) CNT may offer a superior intercalation medium for Li-ions and boost the electrical storage capacity of Li-ion batteries. Hence, the possibility of CNT release in the battery life cycle has to be considered. A case study on CNT suggests that a considerable part of the CNT, utilized in mass consumer products such as batteries and textiles, can be dispersed somewhere in the technosphere or the environment (Köhler, Som et al. 2008).

4.1.5 Vehicle end-of-life

The prevalent method of treating end-of-life vehicles (EOLV) consists of removing vehicle components such as tires, batteries, catalytic converters and large ferrous parts like the engine, transmission and the doors. The remaining body of the car is then shredded into small pieces and most shredder residue is disposed of in landfills or incinerated (Giannouli, de Haan et al. 2007). According to the authors, approximately 75% of the total weight of all EOLV is currently recycled in Europe. The European Union has introduced its End-of-life vehicle Directive in 2000, in order to prevent waste from EOLV and to protect the environment through promoting the collection, re-use and recycling of vehicle components (Gerrard and Kandlikar 2007). According to the Directive, operators must meet recycling targets of 95% by January 2015 (Giannouli, de Haan et al. 2007). Gerrard and Kandlikar (2007) state that currently, 75%-80% of a vehicle is recycled or re-used, which of the majority is ferrous metal. There are strong indications that the EOLV legislation is leading to a reduction in toxic substance use. Carmakers are reducing the number of different plastics being used in order to improve recyclability. It is also likely that the EOLV legislation will increase the use of aluminum, in part due to its ability to be easily recycled. The authors conclude that the next step towards sustainable vehicle management lies in increasing the levels of re-use and remanufacturing (Gerrard and Kandlikar 2007).

Especially in the case of rare earth metals used in vehicle components, recycling is an unsolved issue. For lithium, recycling is almost none existing today, partly due to the low market prices for primary lithium, partly due to the low concentrations in products, making recycling not very attractive (Wäger and Lang 2010). Concerning rare earths and metals of the platinum group, the problem with recycling is similar. Mainly materials from industrial applications are recycled, but not the ones from consumer goods. These products are often globally traded and reused several times, resulting in either no recycling or recycling with low efficiency and high environmental impacts in developing countries (Wäger and Lang 2010). The authors state that for the platinum group metals, about half of the recycling potential is therefore lost annually.

5 LIFE CYCLE INVENTORY

The life cycle inventory (LCI) is structured according to the three main phases of the product life cycle, production, use and disposal. Accordingly, the data to establish the inventory is collected for each phase. Since the focus of this master thesis lies on the materialization of an electric vehicle, the inventory of the production phase has the highest priority. Hence, most time was spent for the data collection for this stage of the life cycle.

The life cycle inventory is constructed in a modular way, allowing various combinations of the established datasets to model different vehicle scenarios (see **Figure 2**).

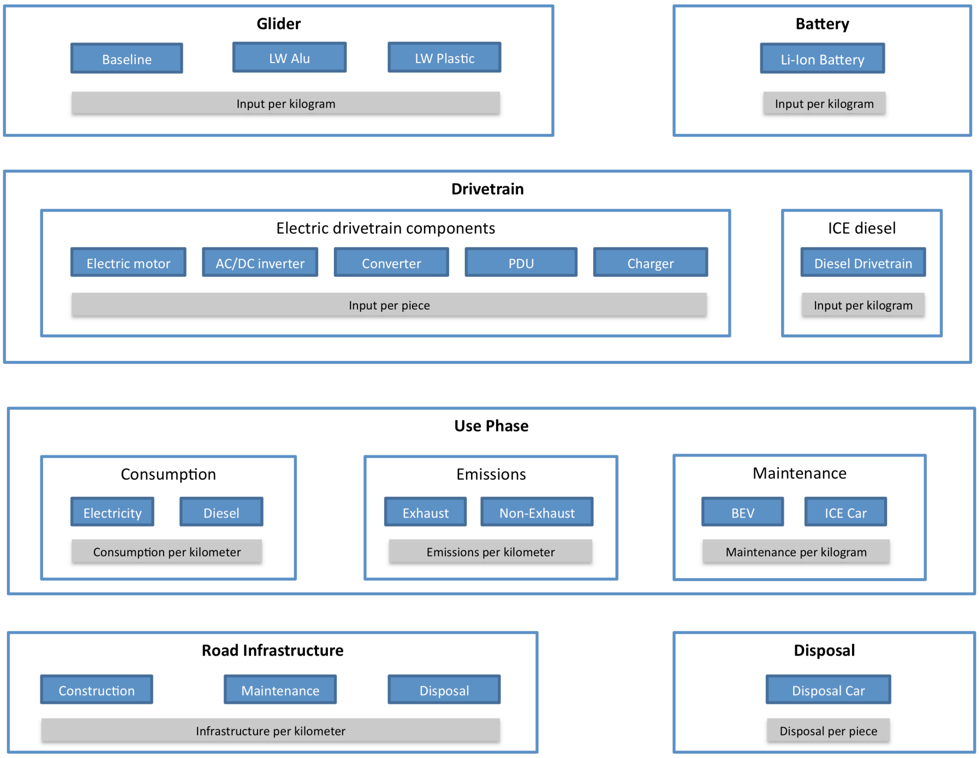

Figure 2 Modularization of the inventory of the product life cycle

It was the goal to create scenarios for three different vehicle classes, which are defined as *mini car, city car* and *compact car*, two powertrain solutions (pure battery electric versus diesel ICE) and three possibilities of glider materialization (1 baseline, 2 lightweight). The combination thereof resulted in 12 different vehicle scenarios, as presented in **Table 2**. In fact, there would be 18 possible combinations of car class, powertrain and glider materialization. However, it was decided to model only the baseline vehicles with a diesel powertrain because the main focus of the study lies on electric vehicles.

Table 2 Overview of vehicle scenarios

	Mini car	City car	Compact car
Baseline	Electric	Electric	Electric
Baseline	Diesel	Diesel	Diesel
Lightweight 1	Electric	Electric	Electric
Lightweight 2	Electric	Electric	Electric

In the following chapters, the created models of the vehicle components that were used to establish the inventory are explained for every phase of the product life cycle. The LCI is calculated for the functional unit of 1 vehicle kilometer.

5.1 Production phase

In the production phase of the LCI, the material content of the modeled cars has to be assessed. This is done separately for the two main compartments of the car, the glider and the drivetrain. It is assumed that the glider is identical for the electric and the conventional car.

5.1.1 Glider

The material content of the glider is modeled based on data from the literature. Unfortunately, it was not possible to acquire more accurate and up-to-date data from manufacturers.

The goal was to model different scenarios of the glider's materialization, one baseline and two lightweight scenarios. The baseline scenario should be representative for the average material content of today's vehicle classes of mini cars, city cars and compact cars. It is assumed that the material composition of cars from these three vehicle classes is very similar. Hence, the same models are applied to all three classes. Further, the majority of BEV to be sold within the next years will probably belong to these classes. Concerning bigger sized vehicles such as vans or cars of the high-end class, the material composition might be significantly different and would have to be modeled separately. Since this is not in the focus of the study, such vehicle types are not considered. The lightweight scenarios should illustrate what a car's material content might look like in the future, if lightweight construction becomes more popular within the next 10 to 20 years.

5.1.1.1 Baseline scenario

The starting point of analysis to construct the baseline scenario was the material content of the Volkswagen Golf A4, as it was modeled by Althaus and Gauch (2010). From this study, the original data used in the LCI was available, which provided a helpful basis. The publication by Althaus and Gauch as well as several other LCA studies of electric mobility are based on the original publication of an LCA of the VW Golf (Schweimer and Levin 2000). Therefore, it was the goal to compare the material content of the VW Golf with similar data from other sources and to create a model with a higher level of detail and a more generic perspective. Unfortunately, there are not many publications on vehicles' material content to be found, which might be the reason why most LCA studies still apply the model of the VW Golf. However, the references listed in **Table 3** give an indication of vehicles' material content and were used to establish the baseline scenario.

Table 3 References on vehicles' material content

Reference	Title
(Weiss, Heywood et al. 2000)	On the road in 2020 - A life-cycle analysis of new automobile technologies
(Tonn, Schexnayder et al. 2003)	An assessment of waste issues associated with the production of new, lightweight, fuel-efficient vehicles
(Giannouli, de Haan et al. 2007)	Waste from road transport: development of a model to predict waste from end-of-life and operation phases of road vehicles in Europe
(Cheah 2010)	Cars on a Diet: The Material and Energy Impacts of Passenger Vehicle Weight Reduction in the U.S.
(U.S. Department of Energy 2010)	Transportation Energy Data Book
(Leduc, Mongelli et al. 2010)	How can our cars become less polluting? An assessment of the environmental improvement potential of cars

The comparison between the references listed in **Table 3** revealed that the materialization of the VW Golf is more or less in line with what these other studies report (see **Appendix I**).

The comparison of the different references on material contents lead to the following baseline model in **Table 4** for the total material content of a car, which can be seen as an average of the data found in the literature. It illustrates the material content of an ICE car without fuel, and the share of each material category is given in percent of the total vehicle weight. The data of the VW Golf A4 (Althaus and Gauch 2010) is added to **Table 4** for comparison.

Table 4 Baseline scenario
Model values of total ICE car material content in percent of the total vehicle weight and values of the Golf A4 for comparison

Material category	Materials	Model values	Golf A4
		% of vehicle weight	% of vehicle weight
Ferrous metals	Steel	63.00%	60.80%
Aluminum	Aluminum	7.00%	5.50%
Non-ferrous metals	Magnesium	0.20%	
	Copper	1.00%	0.70%
	Lead	0.70%	
	Zinc	0.40%	
	Other metals	1.00%	1.80%
Plastics	Plastics	12.00%	16.00%
Synthetics	Glass fiber	0.50%	
	Resins	1.00%	
	Textiles	1.50%	
	Glass	3.00%	2.80%
Rubber	Tires & rubber	4.50%	4.20%
Liquids	Oil & lubricants	2.00%	6.00%
Various	Insulation	1.00%	1.50%
	Paints	1.00%	0.40%
Electronic components	Light emitting diode	0.01%	0.01%
	Printed wiring board	0.19%	0.19%

In order to increase the level of detail, some material classes where further subdivided by specifying single material types (see **Table 5**). The steel fraction is split into conventional steel, stainless steel and high-strength steel (Cheah 2010; U.S. Department of Energy 2010). The aluminum fraction is divided into cast and wrought aluminum (Cheah 2010; Tonn, Schexnayder et al. 2003). The category of plastics is divided into fractions of different specific types of plastics (Giannouli, de Haan et al. 2007; Leduc, Mongelli et al. 2010).

Table 5 Share of single materials per material category

Material	Material specification	Share	References
Steel			(Cheah 2010; U.S. Department of Energy 2010)
	Conventional steel	78%	
	Stainless steel	3%	
	High-strength steel	19%	
Aluminum			(Cheah 2010; Tonn, Schexnayder et al. 2003)
	Cast aluminum	80%	
	Wrought aluminum	20%	
Plastics			(Giannouli, de Haan et al. 2007; Leduc, Mongelli et al. 2010)
	Polypropylene (PP)	51.0%	
	Polyethylene (PE)	15.0%	
	Polyethylene terephthalate (PET)	1.5%	
	Polyamide (PA)	3.0%	
	Polyurethane (PU)	15.0%	
	Acrylonitrile butadine sterine (ABS)	4.5%	
	Other plastics	10.0%	

The model of the glider's material content was complicated by the fact that the cited references do not specify which materials and what amount of a material are used for which component of a car. Usually, the material content is given as the total amount per material in percent of the total vehicle weight, or as the total mass per material in kilogram. Hence, it is not possible to split the material content between glider and drivetrain according to these references. Therefore, it was necessary to apply an assumption about the distribution of materials between drivetrain and glider, which is documented in **Table 6**. Since no other source was available, this assumption is based only on the original data used for the vehicle model (Althaus and Gauch 2010).

Table 6 Assumption on weight shares of materials between glider and drivetrain

Material category	Glider	Drivetrain
Ferrous metals	82%	18%
Aluminum	6%	94%
Non-ferrous metals	32%	68%
Plastics	77%	23%
Synthetics	100%	0%
Rubber	93%	7%
Liquids	14%	86%
Various	100%	0%
Electronic components	100%	0%

The material categories in **Table 4** and **Table 6** are identical, meaning that each material in **Table 4** was assigned to a material category from **Table 6**. Hence, the model values of the total material content from **Table 4** where divided between glider and drivetrain according to the division rules in **Table 6**. This calculation leads to the final model values of the glider's material content for the baseline scenario, which are documented in **Table 7**.

Table 7 Baseline scenario glider
Model values of the material content of the glider in % of the total vehicle weight

Material category	Material	% of total vehicle weight
Ferrous metals	Conventional steel	40.3%
	Stainless steel	1.5%
	High-strength steel	9.8%
Aluminum	Cast aluminum	0.3%
	Wrought aluminum	0.1%
Non-ferrous metals	Magnesium	0.1%
	Copper	0.3%
	Lead	0.2%
	Zinc	0.1%
	Other metals	0.3%
Plastics	Polypropylene (PP)	4.7%
	Polyethylene (PE)	1.4%
	Polyethylene terephthalate (PET)	0.1%
	Polyamide (PA)	0.3%
	Polyurethane (PU)	1.4%
	Acrylonitrile butadine sterine (ABS)	0.4%
	Other plastics	0.9%
Synthetics	Glass	3.0%
	Glass fiber	0.5%
	Resins	1.0%
	Textiles	1.5%
Rubber	Tires & rubber	4.2%
Liquids	Oil & lubricants	0.3%
Various	Insulation	1.0%
	Paints	1.0%
Electronic components	Light emitting diode	0.01%
	Printed wiring board	0.2%
Total Glider		74.91%

In order to model the whole production phase of the glider, further production processes had to be assessed. These include processing of materials, energy expenses as well as emissions. It was assumed that these processes do not vary significantly between different vehicle types. Hence, the data for the process types *auxiliaries* and *emissions* was taken from Althaus and Gauch (2010). Further, it has to be considered that there are material losses in the vehicle production process. The final material amount in the glider is not the same as the material input to the production phase. Therefore, *waste factors* were defined to calculate the total input of material resources for the glider production. Finally, all processes had to be represented by the most appropriate dataset from the Ecoinvent database (see 2.3). The LCI was then calculated per kilogram of glider material, in order to model the differences in weight between the chosen vehicle classes (see **Table 2**). For the documentation of the total inventory of the baseline scenario, see **Appendix II**.

5.1.1.2 Lightweight scenario

The automobile industry has already made attempts to reduce the weight of vehicles in order to increase their fuel efficiency. Lightweight construction might become more widely applied in the future if fossil resources become scarcer and/or if the consumption of fossil fuels gets more expensive due to higher taxes. For electric vehicles, lightweight construction can increase the driving range of the car. Since the battery range depends on both the storage capacity and the vehicle's energy consumption, which in turn shows a linear relationship with the vehicle mass, weight reduction is a strategy to increase range and reduce the energy consumption.

Lightweight construction can be achieved by substituting heavy materials, mainly ferrous metals, by lighter ones. The substituting materials are either light metals, such as aluminum or magnesium, synthetic materials or composite structures, such as carbon fiber or resins. A good example of an electric car that already reflects this strategy is the Tesla Roadster sports car. This luxury electric sports car is built from carbon fiber body panels and its "monocoque chassis, constructed of resin-bonded and riveted extruded aluminum adds rigidity and strength to the lightweight package" (Tesla Motors 2010). This construction allows a vehicle weight of about 1235 kg (including 52 kg motor and 450 kg battery pack) with a driving range of 340 km.

An early effort in the field of lightweight construction was the *Partnership for a New Generation of Vehicles* (PNGV), a research program by the U.S. government starting in 1993 with the participation of some major automobile corporations (Wikipedia 2010b). The main goal of the project was the construction of extremely fuel-efficient vehicles achieving about 80 mpg (corresponding to 3L/100km). In order to reach such a high fuel-efficiency, the vehicle weight should be reduced by 40 percent (Tonn, Schexnayder et al. 2003), and changes in the powertrain system, such as the combination of gasoline engines and electric motors or the introduction of regenerative breaking should be introduced (Wikipedia 2010b). Unfortunately, the Bush Administration has canceled the program in 2001. Nevertheless, some prototype vehicles had been constructed under the PNGV program, for example the General Motors Precept and the Daimler Chrysler ESX2, both diesel-hybrid family cars for five passengers.

The material content of the prototype vehicles of the PNGV program has been analyzed by Tonn, Schexnayder et al. (2003). Based on this reference, two lightweight scenarios were constructed in this study, reflecting the material inventories of the prototype cars *Precept* and *ESX2*. These two vehicles represent different possibilities of lightweight construction. The Precept contains a high share of aluminum substituting heavier materials, whereas the ESX2 is called „composites-intensive", consisting of a high amount of synthetic materials (mainly resins) (Tonn, Schexnayder et al. 2003). The material content of these vehicles was used to model two different lightweight scenarios, a *high plastic* and a *high aluminum* scenario. The first step in the construction of lightweight scenarios for the glider is an inventory of the total material content of each prototype lightweight car. For both vehicles, the storage battery is not included in the material inventory. The total amount of each material is documented in the left-hand column for each prototype vehicle in Table 8. It should be mentioned though that the inventory reflects only a model of these vehicles, based on certain assumptions and approximations.

Table 8 Lightweight scenario glider
Material content of prototype lightweight vehicles ESX2 and Precept

Scenario	Lightweight Plastic		Lightweight Aluminum	
Vehicle name	ESX2		Precept	
Vehicle type	Diesel-hybrid		Diesel-hybrid	
Fuel efficiency	72 mpg		80 mpg	
Total vehicle weight	1021 kg		1175 kg	
Battery weight	40 kg		63 kg	
Materials	**Total amount kg**	**Amount glider kg**	**Total amount kg**	**Amount glider kg**
Ferrous metals	239.5	125.8	220.9	107.2
Cast aluminum	54.4	41.2	371.9	335.8
Wrought aluminum	149.7	113.4	137.9	124.5
Magnesium	55.3	55.3	3.2	3.2
Copper	9.8	7.8	11.1	9.1
Lead	6.9	0.0	7.8	0.0
Zinc	3.9	3.9	4.5	4.5
Other metals	9.8	9.8	11.1	11.1
Titanium	18.1	18.1	15.0	15.0
Platinum	0.005	0.0	0.005	0.0
Plastics	23.6	9.6	84.8	70.8
Polycarbonates	9.1	9.1	0.0	0.0
Glass	31.8	31.8	25.9	25.9
Glass fiber	27.2	27.2	15.9	15.9
Carbon fiber	10.9	10.9	10.0	10.0
Resins	194.1	194.1	39.0	39.0
Textiles	14.7	14.7	16.7	16.7
Tires &rubber	67.1	64.1	34.9	31.9
Liquids	19.6	6.6	22.3	9.3
Insulation	9.8	9.8	11.1	11.1
Paints	9.8	9.8	11.1	11.1
Lights	0.1	0.1	0.1	0.1
Printed wiring board	1.9	1.9	2.1	2.1
All materials	967.2	765.1	1057.3	854.3

Originally, the inventory of materials of the two vehicles in the cited study (Tonn, Schexnayder et al. 2003) contained a category named ‘other materials’, which couldn’t be further specified. Some other materials, which are highlighted in Table 8, were however missing in the study. Hence it was assumed that these missing materials would more or less cover what had been indicated as ‘other materials’, since it seems reasonable that they are used in the construction of the lightweight cars as well. The amounts of these materials (highlighted in Table 8) were calculated under the assumption that their share of the total vehicle weight (without the storage battery) is the same as in the baseline scenario (compare to Table 4). It is argued that the share of the respective materials is likely not to vary substantially between different vehicles from the same class of cars (in this case the golf class). Because of this assumption, the resulting total weight of all materials is not exactly the same as the total vehicle weight minus the battery weight. However, the difference is rather small and is considered not to have a significant influence on the overall lightweight model.

In the next step, the material inventory had to be split between glider and drivetrain, in order to create a material inventory only for the glider. Since it is not possible to identify in which components of the lightweight cars the substitution of heavier materials takes place, it would be wrong to apply the same split factors as in the baseline scenario (see **Table 6**). It is assumed that lightweight construction concerns mainly the glider and less the material content of the powertrain in order to simplify the calculation. Therefore, the material content of the glider was calculated by subtracting the materials of the ICE drivetrain (see 5.1.3) from the total material content. Hence, for every material that is also contained in the ICE drivetrain of the VW Golf A4, the amount of the respective material in the drivetrain was subtracted from the amount in the total material inventory in **Table 8**. The resulting inventory of the glider's material content is documented in the right-hand column for each lightweight scenario in **Table 8**.

It has to be considered though that this assumption might slightly underestimate impacts from the glider in the lightweight scenarios if lightweight construction also affects the drivetrain. On the other hand, the simplified calculation of the lightweight glider weight could also lead to an overestimation of impacts caused by certain materials that are contained to a much higher share in the electric than in the ICE drivetrain. This could for example be the case concerning copper. Since the electric motor contains much more copper than the ICE drivetrain, subtracting the material content of the ICE drivetrain from the total vehicle weight (as in **Table 8**) could lead to an overestimation of the copper content (and respective environmental impacts) of the glider.

So far, the established inventories of the two lightweight options are calculated based on the total weight of the prototype vehicles. Since it was the goal to construct lightweight scenarios for three vehicle classes (mini car, city car and compact car), the inventory had to be scaled to the glider weight of these vehicles in the next step. This step is documented in chapter 5.1.4.

The inventories of both lightweight glider scenarios were completed with further production process types, which are processing, auxiliaries and emissions. The same assumptions were applied as in the baseline scenario to define these processes. The total inventories of both lightweight scenarios in kilograms of glider material are documented in **Appendix IV**.

5.1.2 Electric drivetrain

The life cycle inventory of the electric drivetrain is based on data from the Swiss company Brusa[3] that is specialized on the production of drivetrain components for electric vehicles. The company provided detailed data on the material content of five different components of the electric drivetrain, the electric motor, the AC/DC-inverter, the DC/DC-converter, the power-distribution-unit and the battery charger (see

Table 9).

[3] For more information see: www.brusa.biz

Table 9 Drivetrain components by Brusa

Drivetrain component	Product Brusa	Weight	Specifications
Electric motor	HSM1-6.17.12	53 kg	Hybrid synchronous motor 100 kW
AC/DC-inverter	DMC524	9.5 kg	106 kW
DC/DC-converter	BSC624-12V	4.5 kg	220-450 V, 205 A
Power-distribution-unit (PDU)	PDU	3.9 kg	
Battery charger	NLG5	6.2 kg	

The material content of each drivetrain component was then established based on the information given by the company Brusa (see Table 10). Since the Brusa motor with 53 kg weight and a maximum performance of 100 kW would be rather too big for the mini and city car vehicles, it was scaled down to reflect a lighter and less powerful motor of 31 kg weight and maximum 30 kW performance. The scaling was done based on information on the material content of a permanent-magnet-synchronous motor from the company Perm[4]. The reported material content of the Perm motor in Table 10 does however not represent the exact motor model produced by the company. It can only be interpreted as an approximation.

Table 10 Material content of electric drivetrain components

	Unit	E-motor Brusa	E-motor Perm	Inverter	Converter	PDU	Charger
Aluminum	kg	10.27	8.370	5.05	2.45	1.80	3.10
Brass	kg	0.19	0	0.32	0.06	0.30	0.10
Ceramics	kg	0.15	0.16	0.03	0.03	0.14	0.01
Copper	kg	7.15	5.58	1.43	0.36	0.89	0.72
Ferrite	kg	0	0	0.30	0.51	0	0.76
FR4	kg	0	0	0	0	0	0.26
FR4+Cu	kg	0.05	0.04	0.04	0.28	0	0
FR4+Cu+SMD	kg	0.04	0.04	0.68	0.27	0.47	0.16
NdFeB	kg	1.80	0.93	0	0	0	0
Socketing	kg	0.10	0.09	0.20	0.10	0	0.15
Steel	kg	28.65	13.14	0.32	0.18	0.25	0.33
Stainless steel	kg	5.15	2.36	0	0.10	0	0.19
Synthetics	kg	0.32	0.28	1.13	0.16	0.07	0.44
Total weight	kg	53.8	31.0	9.5	4.5	3.9	6.2

FR4: designation for Flame Retardant glass reinforced epoxy laminate printed circuit boards[5]
SMD: Surface Mounted Device

The energy consumption for the production of the drivetrain components was also included in the inventory, based on information from Brusa. Since the company doesn't conduct the assembly of the motors themselves but gets them assembled from their supplier, information on the production efforts for this process stage was missing. Therefore, the process was approximated with information on the production of an electric motor from ABB (ABB 2002).

Based on the inventories of all components, two different inventories of the electric drivetrain were established, one with the Brusa motor and one with the Perm motor. The other four components are the same in both drivetrain inventories. Additionally, 3 m of copper cable were added to both electric drivetrain inventories to model the connecting materials between the five main components and

[4] For more information see: www.perm-motor.de
[5] For more information on FR4 see: http://en.wikipedia.org/wiki/FR-4

the storage battery. The battery is modeled with the Ecoinvent dataset *battery, LiIo, rechargeable, prismatic, at plant, GLO*. The inventories of each drivetrain component including the choice of Ecoinvent datasets, waste factors and production processes are documented in **Appendix V** The inventories are calculated per piece for each component, since their material content is not scalable in a linear way.

5.1.2.1 NdFeB Permanent magnet

The type of electric motor produced by Brusa is a hybrid-synchronous permanent magnet motor. The permanent magnet is a neodymium-iron-boron magnet with the formula $Nd_2Fe_{14}B$. Since the rare earth metal neodymium is assumed to be critical in terms of environmental impacts (see 4.1.3), it would be important to include the production process of the magnet in the inventory. Unfortunately, there is no Ecoinvent dataset for such permanent magnets. It would be out of scope however to investigate on the detailed production process of permanent magnets within this project. Therefore, a simplified modeling approach was applied to include the permanent magnet materials in the inventory. It was assumed that the magnet is produced from neodymium-oxide, boric-oxide and iron ore. Based on stoichiometric calculations, the required amount of these precursors was calculated. Further, the production efforts were modeled under the assumption that they are similar to the production of primary aluminum. The resulting inventory of the permanent magnet is attached in **Appendix V d**.

5.1.3 ICE drivetrain

The model of the ICE drivetrain was not in the focus of this study. It is needed however to make the results of the LCIA of the electric car comparable to those of a conventional ICE car. Therefore, the same model for an ICE diesel drivetrain was applied as in Althaus and Gauch (2010). The dataset was slightly modified by excluding the fuel from the ICE drivetrain inventory, since including it would result in a double count in the vehicle use phase. Further, the ICE drivetrain was calculated per kilogram to make it scalable for different vehicle sizes.

5.1.4 Vehicle scenarios

Based on the established models of the glider, the electric and the ICE drivetrain, different vehicle scenarios were constructed for the vehicle classes *mini car*, *city car* and *compact car*. The vehicles Smart, Fiat 500 and Volkswagen Golf 6 represent these three vehicle categories. The reason to choose these specific vehicles was that the Smart and the Fiat 500 already exist as electric cars. Smart has started the production of its first series of the *Smart fortwo electric drive*, which is being tested in e-mobility projects in various cities[6]. The Fiat 500 is converted into a BEV by the Swiss company Kamoo, naming it *Kamoo 500 Elektra*[7]. And the VW Golf was included because this study is based on previous work applying the model of a VW Golf (Althaus and Gauch 2010), and because it has already been used in several other LCA studies on electric mobility. Each vehicle class is modeled in a conventional ICE diesel and a pure battery electric version.

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[6] For information on Smart fortwo electric drive see: http://www.smart.ch/information-service-electric-drive/e46d8929-c110-5337-8f7f-9a8c3549a788
[7] For information on Kamoo 500 Elektra see: http://www.kamoo.ch/produkte/kamoo-500-elektra

The diesel vehicles have been modeled only with the baseline glider scenario. The electric vehicles have been modeled with all three glider scenarios. The calculated model values of the vehicle component weights for each scenario are given in **Table 11**. The total vehicle weight includes 75 kg driver weight, since the car's energy consumption in the use phase is calculated based on the vehicle weight including the driver (see 5.2.1).

Table 11 Model values for 12 vehicle scenarios

VEHICLE CLASS		**MINI CAR**	**CITY CAR**	**COMPACT CAR**
Vehicle model		Smart fortwo	Fiat 500	Golf 6
Scenario: ICE Baseline				
Total vehicle weight	kg	845	1055	1314
Glider weight	kg	535	620	838
ICE diesel drivetrain weight	kg	235	360	401
Scenario: BEV Baseline				
Total vehicle weight	kg	889	976	1253
Glider weight	kg	535	620	838
Electric drivetrain weight	kg	55	55	78
Battery weight	kg	224	225	262
Scenario: BEV Lightweight Plastic				
Total vehicle weight	kg	610	652	790
Glider weight	kg	289	335	453
Electric drivetrain weight	kg	55	55	78
Battery weight	kg	191	187	207
Scenario: BEV Lightweight Alu				
Total vehicle weight	kg	616	659	799
Glider weight	kg	294	341	461
Electric drivetrain weight	kg	55	55	78
Battery weight	kg	191	187	208

The total weight of the ICE diesel vehicles corresponds to the weight of the 'existing' vehicles representing each vehicle class[8]. The weight of the glider was assumed to be 74% of the total weight, an assumption based on the weight share of the glider of the Golf 6 (Althaus and Gauch 2010). The weight of the diesel drivetrain results as the difference between drivetrain and total weight. The glider weight of the ICE and the BEV is the same in the baseline scenario in each vehicle class. The battery weight of the electric vehicles has been calculated separately for each scenario, based on the assumed range, storage capacity and discharge rate of the battery (see 5.2.1). The electric drivetrain of the mini and the city car scenario is modeled with the Perm motor, whereas the electric compact car is modeled with the Brusa motor (see 5.1.2). The glider weight of the lightweight cars was calculated based on the assumed weight reduction potential of lightweight construction. For the ESX prototype car a weight reduction of 46% is reported, for the Precept the reduction is reported to be 45%, compared to the standard vehicle models (U.S. Department of Energy 2001). Hence, the glider of the lightweight plastic scenario was calculated to be 46% lighter than the baseline glider for each

[8] For information on vehicle specifications see:
Smart: http://www.smart.at/information-service-downloadcenter/b3a7e3dc-52b2-5dca-a915-2a117d1f4a1e
Fiat: http://www.fiat.ch/dt/will/m136/k367/c682/Preisliste%20500%20August%202008.pdf
Golf: http://www.volkswagen.at/modelle/golf/der_golf/zahlen_fakten/katalog_preise_technische_daten

vehicle class. The glider of the lightweight aluminum scenario was calculated to be 45% lighter than the baseline. These calculations apply the same assumption as in the material inventory (see 5.1.1.2) that mainly the glider is affected by lightweight construction. The weight of the battery and the electric drivetrain are assessed in the same way as for the baseline scenario. The total weight of each lightweight vehicle is finally calculated by summarizing the weights of all components.

5.2 Use phase

Environmental impacts caused during a vehicle's use phase include tailpipe emissions and fuel production, but also non-exhaust emissions, e.g. abrasion from tires. Further, the infrastructure needed to drive a car, such as the construction and maintenance of roads, is included as well. Besides the use of the car, there are also impacts arising from maintaining it, for example due to changing of oil or reparation works.

Of course, the main difference between use phase impacts of electric and conventional cars results from the different power sources and the fact that electric vehicles do not cause any tailpipe emissions. Since this study is conducted from a Swiss perspective, the Swiss electricity mix is applied to power the BEV, consisting of roughly one-third hydropower, one-third nuclear power and one-third imported electricity. The share of electricity from fossil and new-renewable energy sources is very low.

5.2.1 Fuel consumption and exhaust emissions

The fuel consumption and respective tailpipe emissions are calculated for all vehicle scenarios as presented in **Table 11**. The fuel consumption is modeled with the new European driving cycle (NEDC), which is the standard measurement to indicate a vehicles average fuel consumption (Wikipedia 2010a). For the diesel vehicles, the values are taken from the specifications of the vehicle models by the manufacturers[9]. For the electric vehicles, the energy consumption is calculated according to Guzella and Sciarretta (2007) also based on the NEDC, with an overall efficiency of the electric drivetrain of 70%[10]. For the in-town part of the NEDC the efficiency is assumed to be 5% higher because of the effect of recuperation (Althaus and Gauch 2010). The battery weight is calculated iteratively for each vehicle type to achieve a range of 120 km per charge, considering a maximum discharge of 80%. Since the consumption values measured in the NEDC underestimate the fuel consumption in a 'real' driving situation, some correction factors were introduced (Althaus and Gauch 2010). For the ICE diesel vehicles, the consumption values are calculated by adding 13% to the NEDC values. Additionally, 0.6l/100km are added to reflect the fuel consumption for non-propulsion purposes, such as cooling or radio (Althaus and Gauch 2010). For the electric car, the NEDC values are increased by 15% to reflect a real driving situation. Further, 2.7kWh/100km for non-propulsion purposes are added to calculate the total consumption. The resulting total consumption values in l/100km and kWh/100km respectively, as well as the CO_2-emissions of the diesel vehicles in g/km are given in **Table 12**.

[9] For information on vehicle specifications see:
Smart: http://www.smart.at/information-service-downloadcenter/b3a7e3dc-52b2-5dca-a915-2a117d1f4a1e
Fiat: http://www.fiat.ch/dt/will/m136/k367/c682/Preisliste%20500%20August%202008.pdf
Golf: http://www.volkswagen.at/modelle/golf/der_golf/zahlen_fakten/katalog_preise_technische_daten

[10] Efficiency value based on personal communication (A.Krause, Brusa, 2010)

Table 12 Use phase vehicle scenarios
Energy consumption in the NEDC, including correction factors and non-propulsion energy consumption

VEHICLE CLASS		MINI CAR	CITY CAR	COMPACT CAR
Vehicle model		Smart fortwo	Fiat 500	Golf VI
Drag coefficient cw		0.375	0.32	0.31
Drag area A	m2	1.95	2.03	2.22
Rolling resistance coefficient		0.01	0.01	0.01
Scenario: ICE Baseline				
Total vehicle weight	Kg	845	1055	1314
Consumption city	l/100km	4.3	6.6	5.9
Consumption country	l/100km	4.3	4.7	4.4
Consumption total	l/100km	4.3	5.3	4.9
CO_2 emissions	g/km	113.3	146.7	113.1
Scenario: BEV Baseline				
Total vehicle weight	Kg	889	976	1253
Battery weight	Kg	224	225	262
Consumption city	kWh/100km	13.4	14.1	17.0
Consumption country	kWh/100km	19.3	19.1	21.9
Consumption total	kWh/100km	17.0	17.1	19.9
Scenario: BEV Lightweight Plastic				
Total vehicle weight	Kg	610	652	790
Battery weight	Kg	191	187	207
Consumption city	kWh/100km	10.5	10.8	12.3
Consumption country	kWh/100km	17.0	16.3	18.0
Consumption total	kWh/100km	14.5	14.2	15.7
Scenario: BEV Lightweight Alu				
Total vehicle weight	Kg	616	659	799
Battery weight	Kg	191	187	208
Consumption city	kWh/100km	10.6	10.8	12.4
Consumption country	kWh/100km	17.0	16.4	18.0
Consumption total	kWh/100km	14.5	14.2	15.8

To calculate the LCI of the use phase, the energy consumption and exhaust emissions of all vehicle options are modeled with Ecoinvent data. The diesel consumption and exhaust emissions of the ICE vehicles are modeled with the dataset *Operation, passenger car, diesel, EURO5/CH*, which was modified with the specific consumption and CO_2-emission values of each vehicle type. This dataset already includes all non-exhaust emissions. The energy consumption of the BEV is modeled with the Ecoinvent dataset for the Swiss electricity supply mix *Electricity, low voltage, at grid/CH*.

5.2.2 Non-exhaust emissions

Non-exhaust emissions in the vehicle use phase include abrasion from tires and breaking, as well as particles from the road surface. For the diesel vehicles, non-exhaust emissions are included in the operation dataset (see previous paragraph). To model non-exhaust emissions of the electric vehicles, the operation dataset of the ICE vehicles was modified according to Althaus and Gauch (2010) to reflect only emissions from break, road and tire abrasion.

5.2.3 Maintenance

Impacts from maintaining the vehicles in the use phase are modeled as well with Ecoinvent data. For the ICE vehicles, the dataset *Maintenance, passenger car/RER/I* was chosen. It is assumed that maintenance of ICE and electric vehicles is similar, except for processes concerning the drivetrain.

ICE vehicles need regular changes of oil, whereas for EV it is assumed that the battery needs to be replaced once during the vehicle lifetime in 50% of all cases. Therefore, the original Ecoinvent dataset was modified to model maintenance of BEV by excluding emissions of lead and sulphuric acid, which stem from maintenance of the diesel motor. For both ICE and electric vehicles, the maintenance processes are scaled with the vehicle weight, assuming that maintaining bigger vehicles results in higher impacts. Further, it is assumed that the whole maintenance process happens once during the vehicle lifetime. For all three vehicle classes of BEV, half of the battery weight was added to the inventory to reflect the assumed rate of battery replacement.

5.2.4 Infrastructure

The road infrastructure is modeled with Ecoinvent data valid for Switzerland. Infrastructure processes include the construction, maintenance and disposal of roads and are the same for the electric and the ICE vehicle scenarios.

5.3 End-of-Life and Allocation

After establishing the inventories for the production and the use phase of the product system, the disposal phase has to be accounted for as well. However, the inventory of the disposal processes depends on the applied allocation method. In the end-of-life (EOL) phase of the vehicle scenarios, metals are recycled and synthetic materials are subject to waste treatment processes. The recycling of metals generates secondary materials and the waste treatment produces heat and electricity. Hence, the question arises whether these processes belong to the product system. If these EOL processes are accounted to the life cycle of the product, the environmental impacts, but also the recovered materials and energy sources are allocated to the disposed product. However, if it is argued that such processes are not part of the product life cycle, they will not be included in the inventory.

In the basic assessment of the product life cycle, EOL recycling of the materials is modeled with the cut-off approach. The term *cut-off* describes the fact that processes occurring after the first life of the product are cut from the inventory, i.e. not included in the product system. The reason to apply the cut-off approach in the basic model is that in the Ecoinvent database all datasets are modeled with the cut-off approach. Therefore, it is the simplest way to model life cycle impacts of the product, because it is in line with the logic of the Ecoinvent database. The consequence of the cut-off model is that the only EOL processes accredited to the vehicle scenarios are the disposal of those parts of the car and the road, which are not recycled. Recycling processes are however not included in the product system. If material recycling takes place in the EOL phase, the effect on environmental impacts are not accounted to the impacts of the disposed product, but to the next product generation using the recycled materials. Input materials for the production processes of the vehicle scenarios reflect material mixes of primary and secondary materials according to the market situation in the cut-off model.

The argumentation of the cut-off approach can be questioned, since there are different possibilities to deal with the problem of allocation. According to the ISO standard 14044 allocation should be avoided whenever possible through increasing the level of detail or system expansion (ISO 2006b). Therefore, the effect of the allocation approach on the results of the impact assessment will be assessed with a sensitivity analysis (see chapter 7.2), comparing the LCIA results calculated with two different allocation methods. The allocation approach chosen to compare the results of the cut-off model with is allocation through system expansion and substitution. In this approach, recycling and waste treatment processes are accredited to the disposed product. Since these processes generate secondary materials and energy sources, these secondary resources can substitute part of the primary resources. The effect of the substitution of primary materials and energy sources on the total impacts is accredited to the disposed product as avoided impacts. Hence, in the substitution model input materials do not reflect a market mix as in the cut-off, but they are modeled with primary materials, which generate secondary materials through a recycling process at the EOL. Since the Ecoinvent database applies the cut-off approach, it is not possible to model the entire processes affecting the product life cycle with system expansion and substitution. Therefore, it was decided to differentiate between the foreground and the background system (see **Figure 1**). Hence, all datasets of the foreground system are adapted to the substitution model, whereas the background system remains unchanged.

The difference between the cut-off and the substitution model is illustrated in **Figure 3** and **Figure 4**. In **Appendix III**, the inventory of the baseline glider is documented in the substitution model. It shows the differences of the LCI between the two allocation approaches. Hence, the inventory in **Appendix III** is meant as an example for all other vehicle components, which have all been modeled with system expansion and substitution according to the same logic.

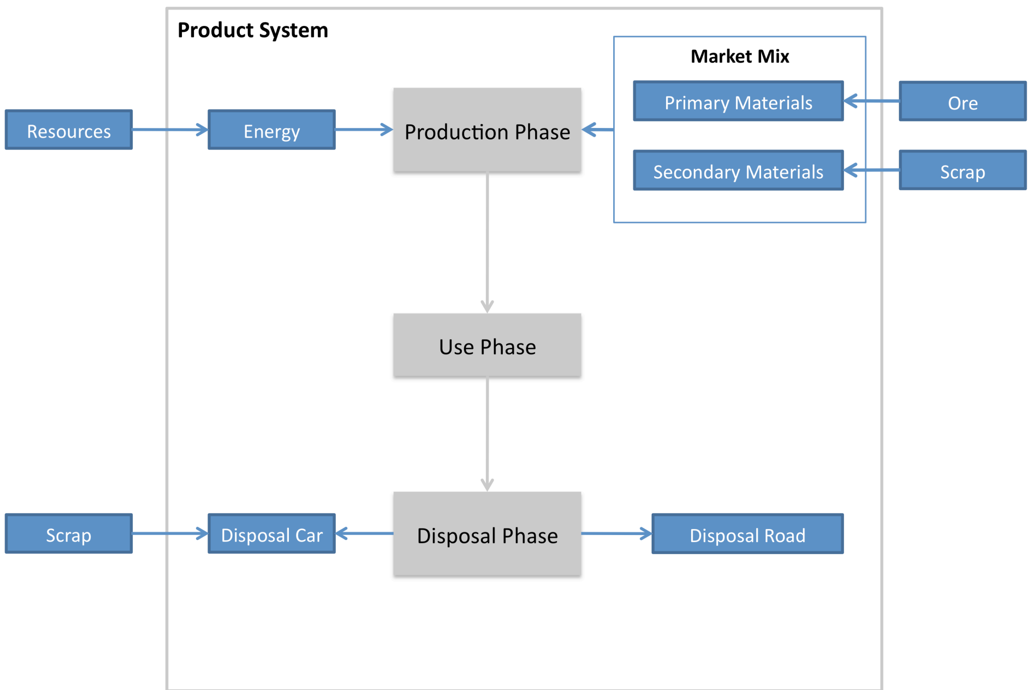

Figure 3 Cut-off model of product system

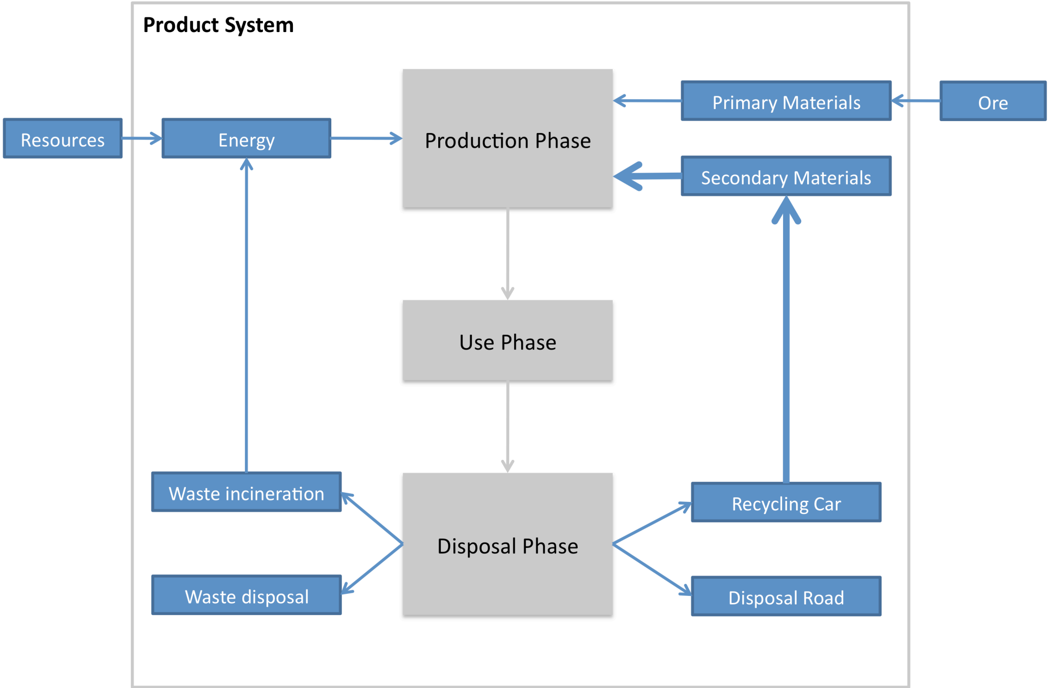

Figure 4 Substitution model of product system

6 IMPACT ASSESSMENT

In this chapter, the results of the life cycle impact assessment of the vehicle scenarios are presented. All results have been calculated on the basis of the inventory of the product life cycle explained in chapter 5. The environmental impacts of the vehicle scenarios are assessed only for the cut-off model, and they are calculated for several impact categories. The results are first presented separately for each impact assessment method (chapter 6.2) and then in a comparison of all applied methods (chapter 6.3). Finally, the LCA results are compared to the results of the study by Althaus and Gauch (2010) (chapter 6.4). All results presented in chapter 6 will then be interpreted and discussed in more detail in the interpretation in chapter 8.1.

6.1 Impact assessment methods

The following impact assessment methods have been chosen to analyze the environmental impacts of the vehicle scenarios:

Ecoindicator (EI 99): Environmental impacts are calculated with ecoindicator 99. Impacts are assessed without weighting by the characterization results of the endpoint categories human health, ecological quality and resource quality. Total impacts are also calculated in total ecoindicator scores (EI-pt), which are given as the weighted and aggregated endpoints. The higher the EI score, the higher are the environmental impacts.

Ecological scarcity (UBP 06): Ecological scarcity is applied as another method to assess environmental impacts with an aggregated total score, given in ecopoints[11]. Again, a high score describes a high impact.

The analysis with ecoindicator 99 and ecological scarcity should allow for a good interpretation and comparison of the total environmental impacts of each scenario. However, applying weighted single score indicators is not allowed for comparative assertions according to the ISO standard. It was decided to use these indicators nevertheless, because single score results are easier to communicate and provide a good overview of the assessment. In contrast, the ecoindicator endpoint categories focus on specific types of environmental impacts. Since they are not weighted, their application is compatible with the ISO standard. The assessment methods ecoindicator and ecological scarcity were chosen because they are very popular in Switzerland, regarding that the study is conducted from a Swiss perspective.

Those categories of impacts that are thought to be most relevant for the life cycle of electric vehicles are analyzed specifically with selected single indicators. It is assumed that the global warming potential, the cumulative energy demand and the demand for metal resources are the most important issues in terms of impacts of BEV. Therefore, the following single indicators were chosen to analyze these specific impact categories:

Global warming potential (GWP): The impact assessment method IPCC 2007 calculates the GWP in kilograms of CO_2 equivalents (kgCO_2-eq). The time perspective of 100 years was chosen for the assessment.

Cumulative energy demand fossil (CED fossil): This single indicator assesses the demand for fossil energy resources in mega joules equivalents (MJ-eq).

[11] ecopoints = UBP = Umweltbelastungspunkte

Cumulative energy demand nuclear (CED nuclear): This single indicator assesses the demand for nuclear energy resources in mega joules equivalents (MJ-eq).

Cumulative energy demand total (CED total): The indicator for the total energy demand includes energy from fossil and nuclear resources, biomass from renewable and non-renewable sources as well as the renewable resources solar, wind, geothermal and hydropower. The CED total score is also calculated in mega joules equivalents (MJ-eq).

Cumulative exergy demand metals (CExD metals): This single indicator assesses the demand for metallic resources according to their exergy content in mega joules equivalents (MJ-eq).

6.2 Results

The results of the impact assessment of the 12 vehicle scenarios in the cut-off model are presented separately for each impact category. The LCIA was calculated in relation to the functional unit one *vehicle kilometer* (vkm) for every indicator. The impacts are split between the main phases of the vehicle life cycle as well as between the main vehicle components. The use of labels in the graphical presentation of the LCIA results is defined in **Table 13**.

Table 13 Declaration of labeling of impact assessment results

Labels	Processes
Road	Construction road + maintenance road + disposal road
Glider	Production of glider
Drivetrain	Production of drivetrain
Battery	Production of battery + battery replacement
Energy & Emissions	Energy consumption + exhaust emissions + non-exhaust emissions
Maintenance car	Maintenance of car (without battery replacement)
Disposal car	Disposal of car

The figures presenting the LCIA results are structured in the same way for all impact categories. The vehicle scenarios are ordered first according to the vehicle class (mini cars, city cars, compact cars) and secondly, according to the glider scenario (baseline, lightweight plastic, lightweight aluminum). In each vehicle class, the ICE baseline scenario is presented next to the BEV baseline scenario. This arrangement should make it possible to compare conventional against electric vehicles, to analyze the impacts of different vehicle classes, as well as the impact of the glider materialization within each vehicle class.

The processes depicting the road and the disposal of the car have been modeled equally in all vehicle scenarios. Therefore, their impacts are always the same in all scenarios and their contribution varies only depending on the assessment method. The impacts of the battery, the maintenance of the car and energy and emissions vary only depending on the weight of the battery and the vehicle. Therefore, they are highest for the baseline EV in all impact categories, because the three baseline vehicles have the heaviest batteries, the highest total vehicle weight and also the highest energy consumption in the use phase of all BEV scenarios. Hence, the baseline compact car always has the highest impacts in these three categories in all impact assessment methods.

6.2.1 Ecoindicator human health

Impacts on human health of the vehicle scenarios are presented in Figure 5, calculated in DALY/vkm (DALY=Disability Adjusted Life Years Lost).

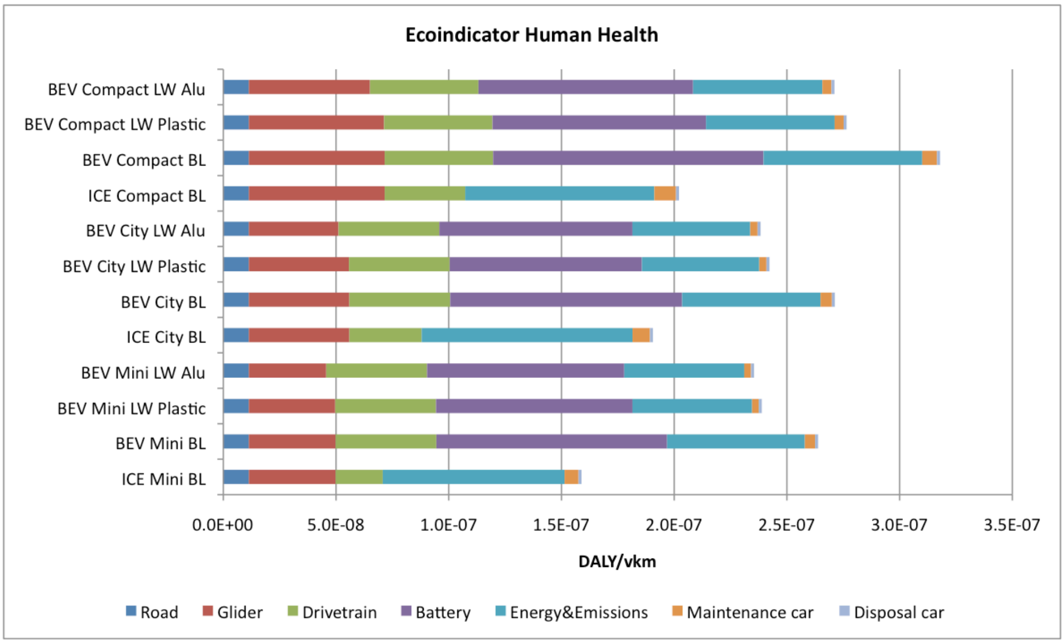

Figure 5 Impacts on human health of the vehicle scenarios

Comparing the impacts on human health within the baseline scenario, the electric vehicles have higher impacts than the ICE cars in all three vehicle classes. The highest contribution to the impacts of the electric vehicles has the battery with about 40% of the total score of each BEV scenario. For the ICE cars, the energy consumption and emissions have the highest impact with 40%-50% of the total score, whereas for the BEV impacts from energy consumption contribute 23% to the total impacts. Glider and drivetrain together are responsible for about half of the impacts of the ICE vehicles, whereas for the electric vehicles the share is about 30%. The drivetrain of the diesel vehicle has 25%-50% lower impacts on human health than the electric drivetrain without the battery.

Regarding the BEV scenarios, the baseline scenario has the highest impacts on human health in every vehicle class. The two lightweight scenarios have very similar total impacts in each vehicle class. The compact car vehicles have the highest impacts within each materialization scenario, whereas the city and the mini cars are similar within the materialization scenarios. The lightweight aluminum glider has the lowest impacts, whereas the lightweight plastic and the baseline glider have the same impact score. The contribution of the glider ranges from 15%-22%, those of the drivetrain from 15%-19%. In all nine BEV scenarios, the battery contributes the most to human health impacts with 35%-40%, followed by the energy consumption with 21%-23%.

6.2.2 Ecoindicator ecosystem quality

Impacts on ecosystem quality are calculated as the potentially disappeared fraction per vehicle kilometer (PDF*m^{2}yr/vkm). The resulting impact scores of the vehicle scenarios are shown in **Figure 6**.

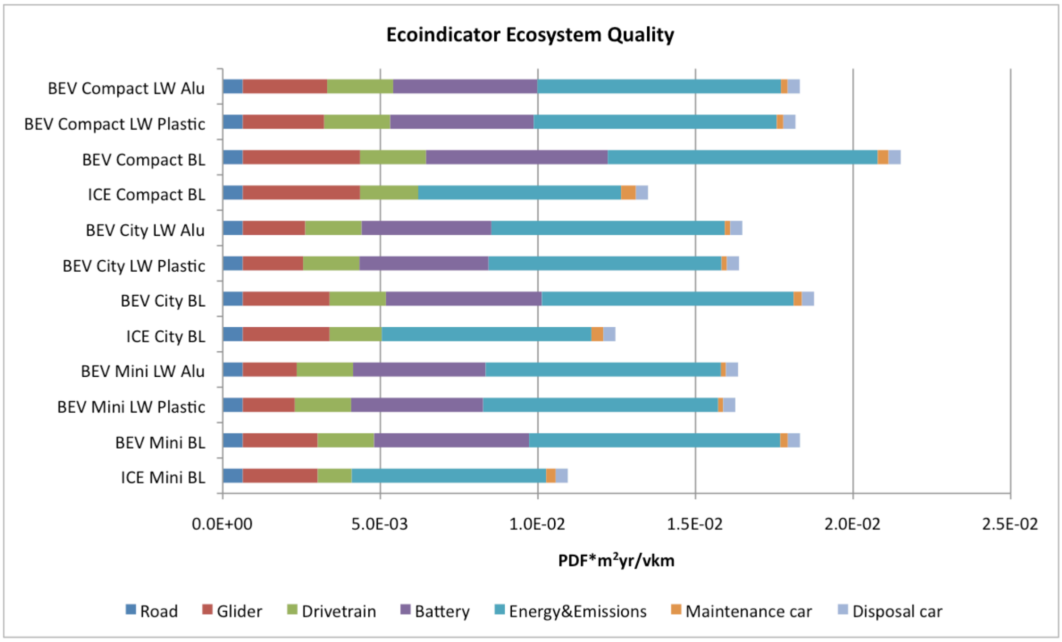

Figure 6 Impacts on ecosystem quality of the vehicle scenarios

In the baseline scenario, the electric vehicles have 30%-40% higher impacts on ecosystem quality than the ICE vehicles in each vehicle class. The highest impacts result from the energy consumption for all six scenarios, which contributes 40%-44% for the BEV, and 48%-56% for the ICE cars respectively. The electric drivetrain has higher impacts than the diesel drivetrain.

Comparing the electric vehicle scenarios, the energy consumption and emissions have the highest contribution (40%-46%) to the total impacts on ecosystem quality. Second highest contribution has the battery with 25%-27%. However, the impacts caused by the glider have the biggest variation between the different BEV scenarios. They are highest for the baseline compact car and lowest for the lightweight plastic mini car. Generally, the compact car scenario has the highest impact and the mini and city car scenarios are very similar if modeled with the same glider materialization. Both lightweight scenarios result in lower impacts than the baseline scenario in each vehicle class.

6.2.3 Ecoindicator resource quality

Impacts on resource quality are calculated in mega joules surplus per vehicle kilometer (MJ surplus/vkm). The results are presented in **Figure 7**.

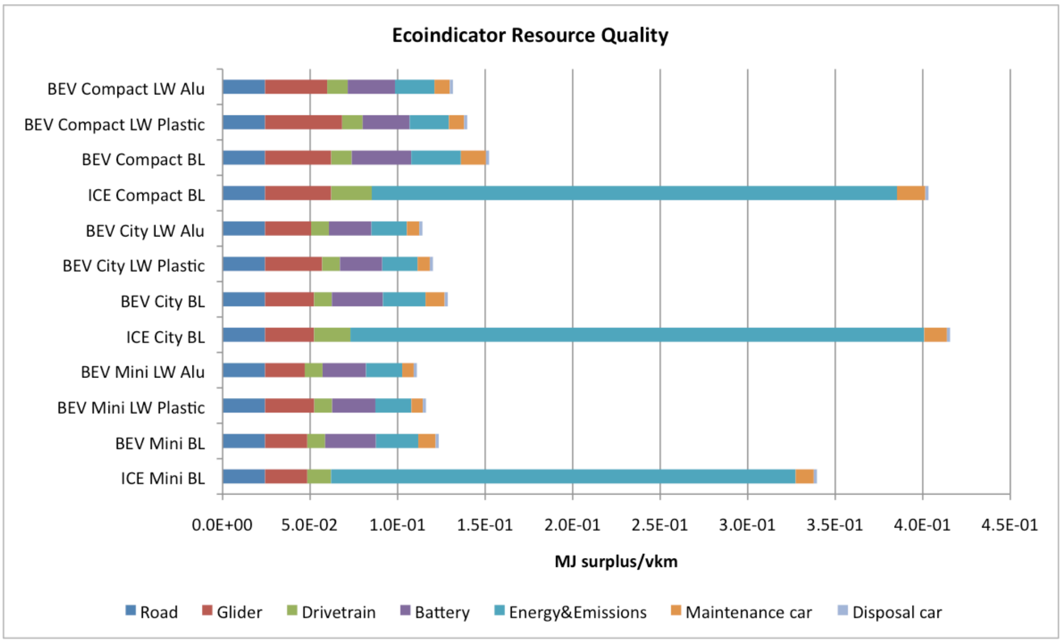

Figure 7 Impacts on resource quality of the vehicle scenarios

Regarding impacts on resources, the impact of the ICE car is almost three times higher compared to that of the electric car. The main difference lies in impacts from the energy consumption, which contribute 75%-80% to the total impacts of the ICE vehicles. For the electric vehicles, the contribution of energy consumption and emissions is only about 20% or more than ten times lower than for the diesel vehicles. The ICE drivetrain causes higher impacts than the electric drivetrain.

The compact car scenarios have the highest impacts on resources, comparing all electric vehicles. In each vehicle class however, the lightweight aluminum scenario has the lowest and the baseline scenario the highest impacts. The impacts from the glider and from maintenance vary the most between the different BEV scenarios. The glider of the compact car vehicles causes 20%-30% higher impacts than the one of the other vehicle classes. Within each vehicle class, the lightweight plastic glider has the highest and the lightweight aluminum the lowest impact score. The contribution of the glider to the total score is 20%-32%. All other processes have similar impacts in all BEV scenarios.

6.2.4 Ecoindicator single score

The results of the assessment with ecoindicator are now presented in total ecoindicator scores (EI-pt) in Figure 8, which are given by the weighted and aggregated endpoint scores (see 6.2.1-6.2.3). The hierarchic perspective was applied for the choice of weighting factors (EI 99 H/A).

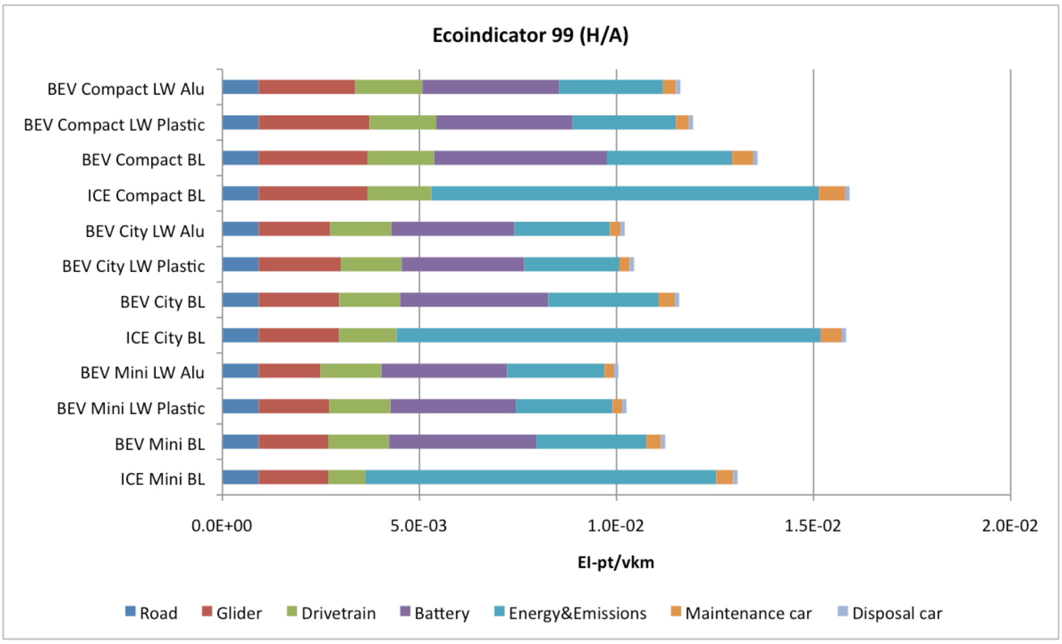

Figure 8 Impacts of the vehicle scenarios assessed with ecoindicator 99

For all three vehicle classes, impacts are 10%-30% higher for the ICE car than for the BEV in the baseline scenario. Impacts of the ICE vehicles result mainly from energy consumption and emissions in the use phase, which contribute 62%-68% to the total impact score. For the BEV, the energy consumption has a contribution of about 25%, which is two to three times lower than that of the ICE vehicles. The highest contribution to the total impacts of BEV originate from the battery with roughly one-third of the total impact score for all three baseline scenarios of electric vehicles. The contribution of the glider to the total impacts varies from 13%-20%, the one of the drivetrain from 7%-10% for the ICE vehicles and from 12%-14% for the electric vehicles. The electric drivetrain causes slightly higher impacts than the diesel drivetrain.

Comparing the BEV scenarios, the compact car baseline scenario has the highest total impacts. The three compact car scenarios have higher total impacts than all other BEV scenarios. In each vehicle class, the baseline scenario has the highest score, whereas the lightweight aluminum scenario has the lowest score. The differences in impacts between the scenarios depend mainly on the contribution of the glider, which varies between 16% and 24%. In each vehicle class, the lightweight plastic glider has the highest impacts and the lightweight aluminum the lowest. Overall however, the heavier glider components of the compact cars cause higher impacts than those of the smaller vehicle classes, irrespective of the materialization. Further, the battery has an important influence on the total impacts, since it has the highest contribution with about 30%.

6.2.5 Ecological scarcity

The assessment of ecological scarcity is presented in Figure 9. The impacts are calculated as weighted and aggregated endpoint scores, in ecopoints per vehicle kilometer.

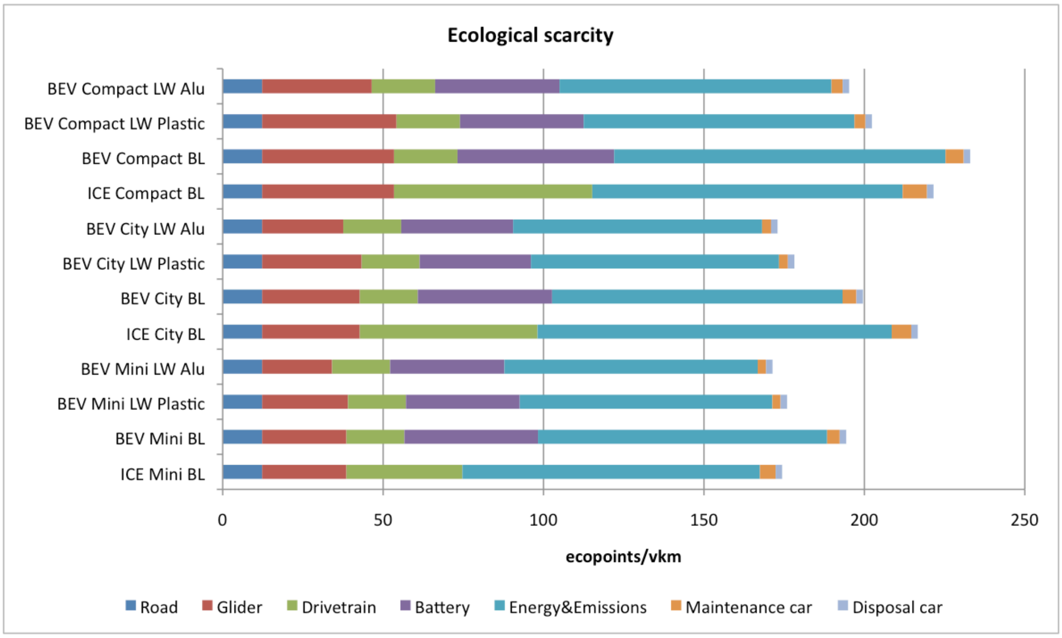

Figure 9 Impacts on ecological scarcity of the baseline vehicle scenarios

In the comparison of the ICE and the BEV baseline scenarios assessed with ecological scarcity, the electric car has slightly higher impacts than the ICE car in the compact and the mini car class, whereas in the city car class the electric vehicle has slightly lower impacts. For both the conventional and the electric vehicles, the use phase causes a high share of the total impacts. The contribution of energy and emissions is about 45% for the BEV. For the ICE vehicles it ranges between 44% and 53%. In the mini car class, impacts from energy and emissions are almost equal for both conventional and electric car. In the city car class however, the impacts of the ICE car are about 20% higher in this category. Impacts from the ICE drivetrain are twice to three times as high as those of the electric drivetrain. Impacts from the glider contribute 14%-18%, depending on the vehicle size.

Total impacts of the BEV scenarios depend mainly on the energy consumption and emissions. In each vehicle class, the baseline scenario has 10%-15% higher impacts than the two lightweight scenarios, which show very similar results. The lightweight aluminum glider has 10%-15% lower impacts than the other two glider scenarios, which have similar scores in each vehicle class. The battery contributes roughly 20% to the total score of all electric vehicle scenarios.

6.2.6 Global warming potential

The assessment of the GWP of the vehicle scenarios is presented in **Figure 10**. The results are calculated in kilograms of CO_2 equivalents per vehicle kilometer, with a time perspective of 100 years.

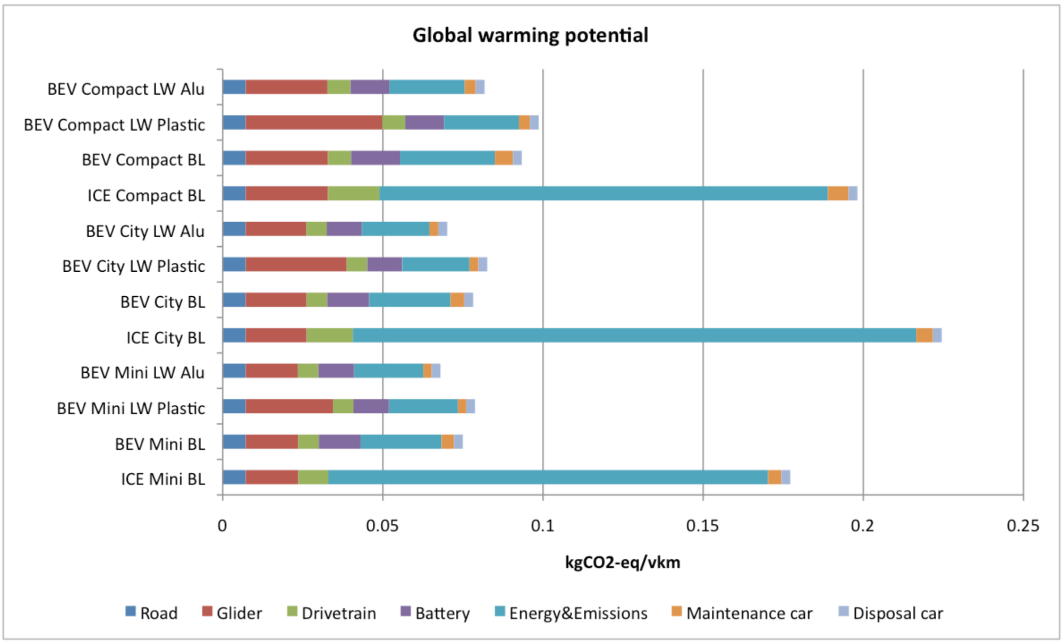

Figure 10 Impacts on GWP of the vehicle scenarios

Comparing the GWP between the baseline scenarios, the total GWP of the ICE car is about twice as high as that of the electric car in each vehicle class. The difference is mainly due to the impacts from energy consumption and emissions, which are 60%-85% higher for the ICE vehicles. Use phase impacts contribute 32%-34% to the total GWP of the BEV scenarios, and 71%-78% to the GWP of the ICE scenarios. The ICE drivetrain causes higher impacts than the electric. Impacts of the electric drivetrain are 20% lower in the mini car class and 50% lower in the city and compact car classes.

In the comparison of the GWP between the electric vehicle scenarios, the lightweight plastic scenario has the highest impacts, whereas the lightweight aluminum scenario has the lowest impacts within each vehicle class. The lightweight plastic glider has 30% higher impacts than the other two glider scenarios, which result in very similar scores. Concerning energy and emissions however, the baseline scenario has 10%-20% higher impacts than the other two. In all three lightweight plastic scenarios, the glider contributes the most to total GWP with 35%-43%, whereas in all baseline scenarios, the energy consumption has the highest contribution with 32%-34%. For the lightweight aluminum scenarios, the split of contributions depends on the vehicle size. Comparing between vehicle classes, the compact car class has the highest and the mini car the lowest impacts within each materialization scenario.

6.2.7 Cumulative energy demand fossil

The demand of fossil energy sources is calculated in mega joules equivalents per vehicle kilometer and the results are shown in Figure 11.

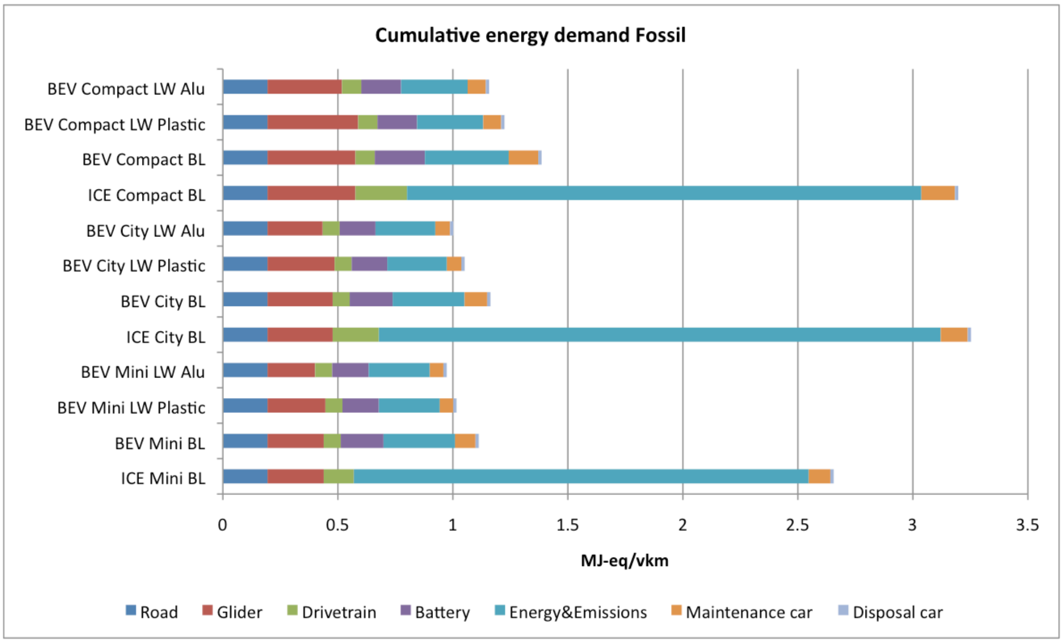

Figure 11 Impacts on CED fossil of the vehicle scenarios

The cumulative fossil energy demand of the ICE car is twice to three times as high as for the BEV, depending on the vehicle class. The energy consumption and emissions are dominating in the ICE scenarios, contributing 70%-75% to the total score. The ICE drivetrain causes a two to three times higher demand for fossil energy sources than the electric drivetrain.

Comparing the fossil energy demand of the BEV scenarios, the baseline scenario reaches the highest and the lightweight aluminum scenario the lowest score within each vehicle class. Concerning impacts of the glider however, the lightweight plastic glider has the highest impacts, whereas the lightweight aluminum has the lowest. The contribution of the glider varies from 21% to 32%. The glider has the highest share of the total score in the compact class, whereas in the mini class the energy consumption and emissions have the highest share. In comparison to the other impact assessment methods, the road has quite a high contribution to the total fossil energy demand with 14%-20%.

6.2.8 Cumulative energy demand nuclear

The impacts on the nuclear energy demand are presented in Figure 12, calculated as well in mega joules equivalents per vehicle kilometer.

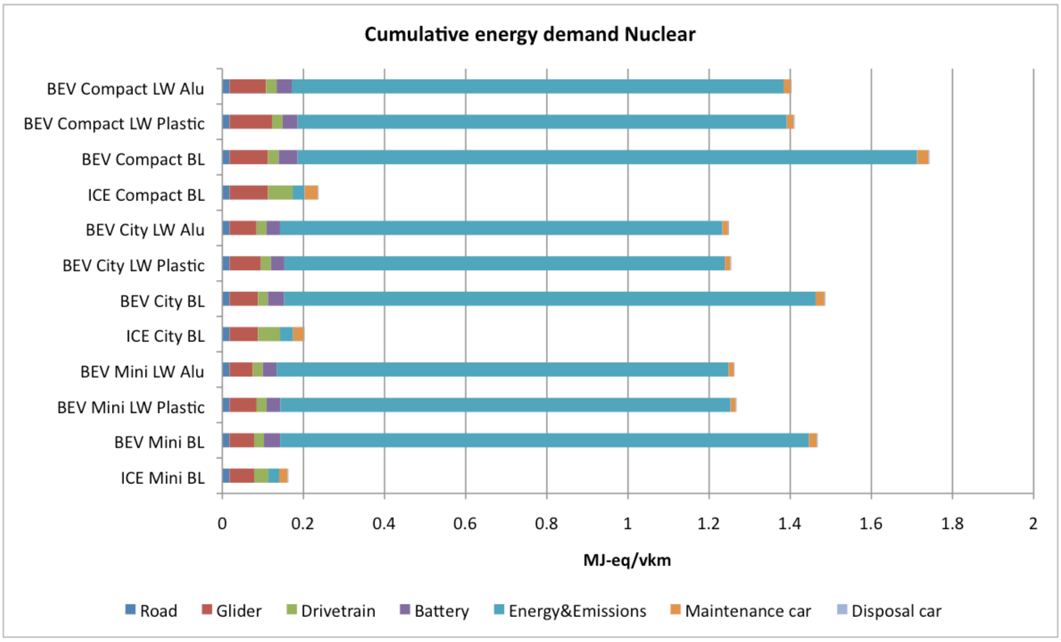

Figure 12 Impacts on CED nuclear of the vehicle scenarios

The cumulative nuclear energy demand is strongly dominated by the energy consumption in the use phase of the BEV scenarios, which has a share of about 90% of the total score. Since the respective demand of the ICE vehicle scenarios is very small, the total nuclear energy demand of the electric vehicles is much higher. For all other processes, the demand is quite low in all scenarios.

Comparing the BEV scenarios, the baseline car has the highest contribution, whereas the two lightweight cars are very similar in each vehicle class. Regarding the materialization scenarios, the mini and the city car classes are very similar as well. The compact car class has a slightly higher nuclear energy demand within each materialization scenario, however.

6.2.9 Cumulative energy demand total

The total cumulative energy demand of the vehicle scenarios is presented in **Figure 13**, calculated in mega joules equivalents per vehicle kilometer.

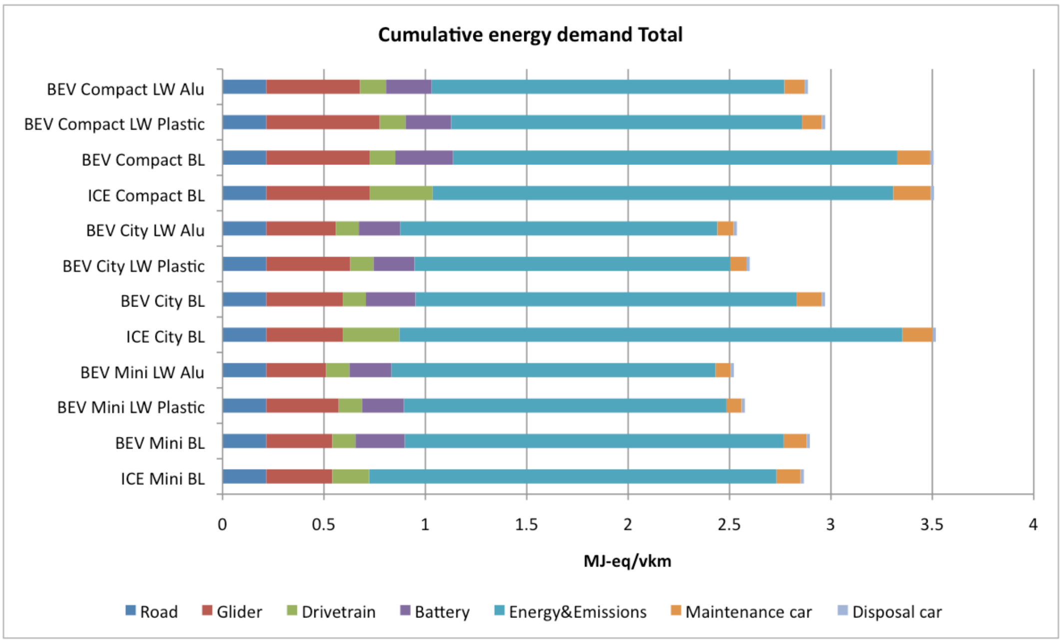

Figure 13 Impacts on CED total of the vehicle scenarios

The total energy demand is the same for the ICE car and the BEV of the compact car class in the base-line scenario. In the city car class, the ICE vehicle has the higher total demand, whereas in the mini car class, the BEV has a slightly higher energy demand. The total score is dominated by impacts from energy and emissions. The diesel drivetrain has a higher total energy demand than the electric drivetrain.

Energy consumption and emissions have a share of about 65% of the total cumulative energy de-mand in all BEV scenarios. In each vehicle class, the baseline scenario has the highest and the light-weight aluminum scenario the lowest impacts. Concerning the glider however, the lightweight plastic glider causes the highest CED.

6.2.10 Cumulative exergy demand metals

The cumulative exergy demand for metals is presented in Figure 14 calculated in mega joules equivalents per vehicle kilometer.

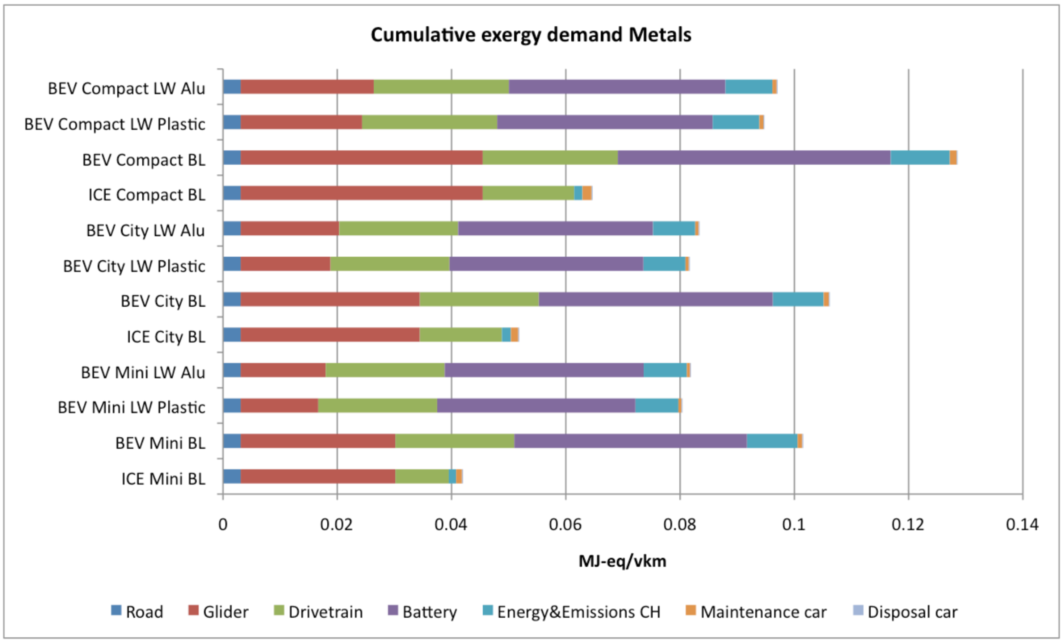

Figure 14 Impacts on CExD metals of the vehicle scenarios

Comparing the ICE and the electric cars in the baseline scenario, the exergy demand for metals of the BEV is twice as high. For the conventional vehicles, the glider contributes the most to the total demand with 60%-65% of the total score. In the BEV scenarios, the battery has the highest contribution of roughly 40%. The ICE drivetrain has 30%-50% lower impacts than the electric drivetrain.

Comparing only the electric cars, the baseline scenario has 20%-25% higher impacts than the two lightweight scenarios, which show very similar results in each vehicle class. The battery contributes the most to the total impacts of the electric vehicles. The use phase of the car has a small effect on the CExD for metals.

6.3 Comparison of impact assessment results

In chapter 6.2 the results of the impact assessment have been presented separately for each impact assessment method. With this structure it is well possible to compare the environmental impacts of the different vehicle scenarios within one assessment method. However, it is quite difficult to analyze and compare the overall performance of each vehicle scenario from a general point of view. Therefore, the results of the LCIA are now presented again in a comparison between all impact assessment methods that have been applied.

The assessed impacts of each vehicle scenario are calculated in relation to a reference scenario, which is defined to be the BEV compact car baseline scenario. Hence, the impacts of the reference scenario are set to 100% and the impact scores of all other scenarios are calculated as a percentage of the score of the reference scenario. The resulting comparison of the vehicle scenarios is presented in Figure 15 and Figure 16. Figure 15 only compares the six baseline scenarios, in order to compare the electric vehicles against conventional diesel cars in different vehicle classes. In Figure 16, the nine

BEV scenarios are compared against each other to analyze the effects of the differences in glider materialization, vehicle weight and vehicle size.

The comparison of all impact assessment results shows that there is neither one scenario having the highest impacts nor one scenario having the lowest impacts throughout all assessment methods. The reference scenario has higher impacts than all other electric vehicle scenarios for all indicators, except for the GWP of the BEV compact car lightweight plastic scenario. The GWP of this scenario is about 5% higher than the reference case's. The variation of the impact assessment results depending on the type of environmental impacts assessed is generally much higher for the ICE vehicle scenarios than for the BEV scenarios.

Figure 15 shows the comparison of the BEV and the ICE vehicles only in the baseline scenario. It can be seen that the electric mini and city cars have about 10%-20% lower impacts than the reference throughout all indicators. The variation between the results of the different assessment methods is very low for the BEV scenarios. However, the ICE scenarios have two to almost three times higher impacts than the reference with respect to the indicators EI resource quality, CED fossil and GWP. Concerning the EI total score, the ICE city and compact car scenarios have about 10% higher impacts than the reference. The ICE mini car however has slightly lower impacts than the reference case. In contrast, the ICE scenarios have much lower impacts with respect to certain other indicators. In the categories EI ecosystem quality and human health, the impacts of the ICE scenarios are 40%-50% lower compared to the reference. Concerning the indicator CExD metals, the impacts are 50%-60% lower and with respect to CED nuclear they are 80%-90% lower.

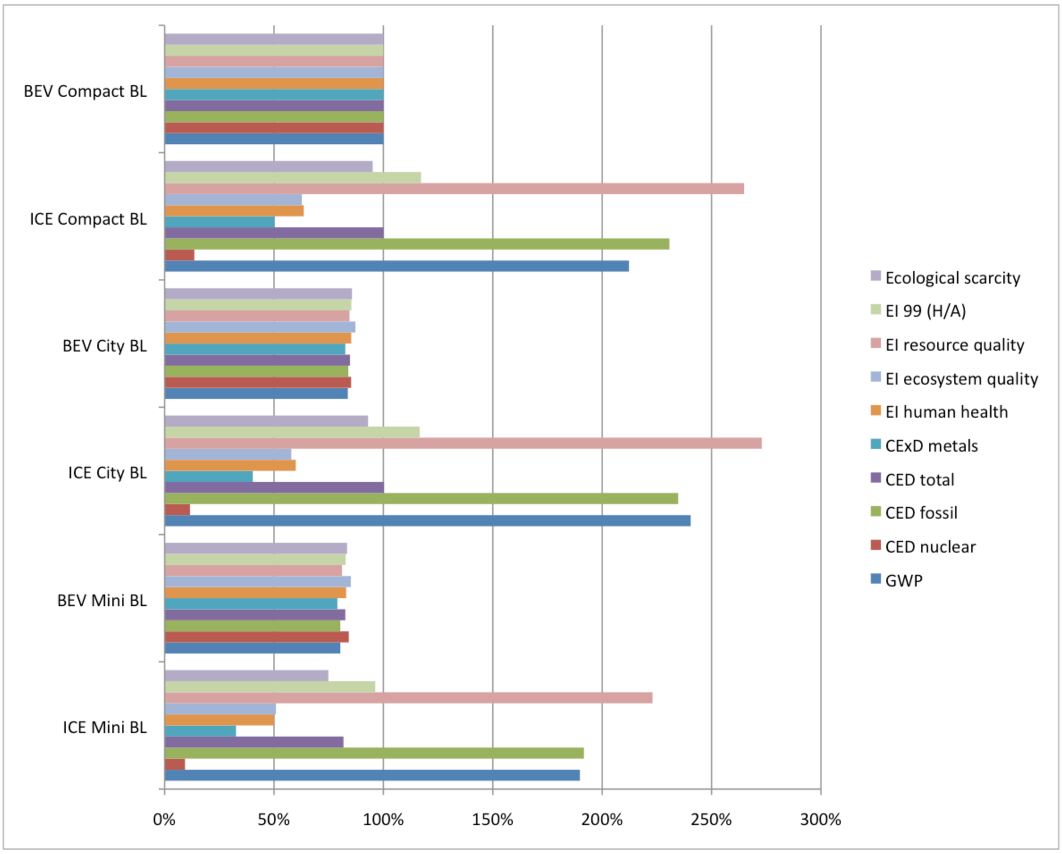

Figure 15 Comparison of results of ICE and electric vehicles in the baseline scenario

To conclude on the comparison in **Figure 15**, the ICE and BEV scenarios are compared within each vehicle class. In the compact car class, the electric car has higher impacts than the ICE car concerning ecological scarcity, EI ecosystem quality and human health, CExD metals and CED nuclear. It has lower impacts with respect to EI resource quality and EI total score, CED fossil and GWP, and the scores for CED total are equal. The same conclusion holds true for the city car class, except for ecological scarcity and CED total, which are lower for the BEV. Concerning the mini car class, the comparison of impacts between ICE and BEV depending on the assessment method leads to the same conclusions as for the compact car class. Generally speaking, it depends on the type of impact category whether the electric or the ICE diesel vehicles have lower environmental impacts. However, it is always the mini car that achieves the best result. In the baseline scenario, the ICE mini car has the lowest impacts in six impact categories, whereas the electric vehicle shows the best result in four impact categories.

In **Figure 16**, the results of the electric vehicle scenarios baseline, lightweight plastic and lightweight aluminum are compared between all impact assessment methods. Again, the order of best and worst case scenario depends on the type of impact category. The variation between the different scenarios is however much smaller than in the comparison of ICE and BEV scenarios.

The reference case, compact car baseline, has the highest impacts for all indicators, except for the GWP, which is about 5% higher for the compact car lightweight plastic. The lowest GWP achieves the mini lightweight aluminum with a 30% lower score compared to the reference. This scenario also has the lowest impacts concerning the impact categories CED fossil, EI human health and resource quality as well as total EI score, with a 25% to 30% reduction compared to the reference. The mini lightweight plastic scenario has the lowest CExD for metals (37% lower than the reference). With respect to the remaining indicators there are two or more scenarios achieving the maximum reduction of impacts compared to the reference case. Generally, it can be differentiated between impact categories, which are highest for the biggest vehicle class, irrespective of the materialization scenario, and such which are highest for the baseline scenario, irrespective of the vehicle size. The three compact car scenarios have the highest impacts in the categories EI resource quality and total EI score. In contrast, the compact, city and mini car baseline scenarios have the highest impacts concerning CED nuclear and CExD metals. Within each vehicle class, both lightweight scenarios have lower impacts than the baseline in all impact categories, except for GWP. This indicator is always about 5% higher for the lightweight plastic scenario.

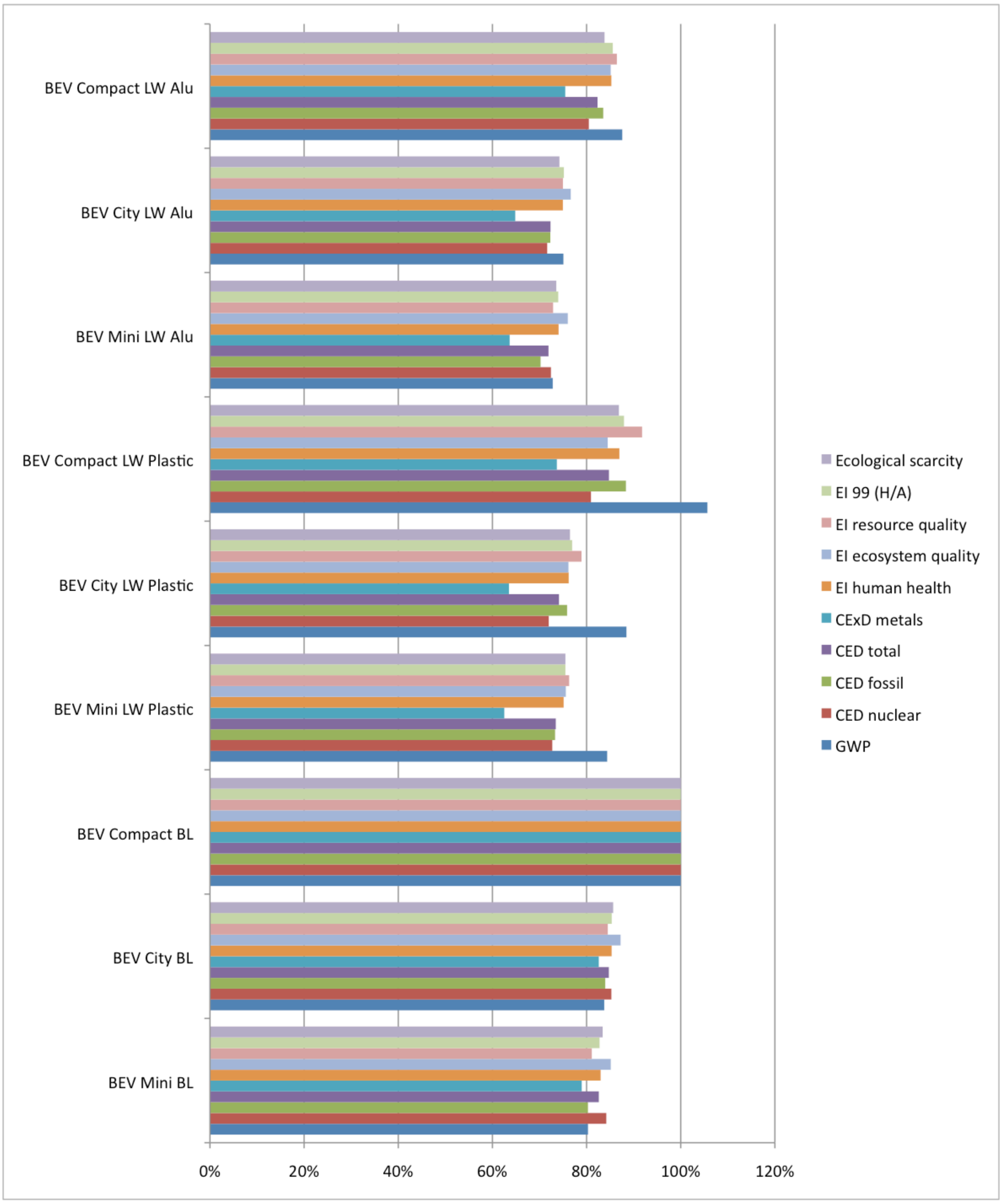

Figure 16 Comparison of results of BEV scenarios

6.4 Comparison of results with Althaus and Gauch (2010)

Since the baseline model of the compact class electric vehicle was based on previous work by Althaus and Gauch (see paragraph 5.1.1.1), the results of the impact assessment of the BEV compact car baseline scenario are now compared with the impact assessment of the electric passenger car model from this former study (Althaus and Gauch 2010). The two vehicle models differ slightly in their composition of glider materials, but mainly in the model of the electric drivetrain components. Further, the storage battery of the vehicle by Althaus and Gauch is considerably heavier with a wei ght of 400kg. Both models apply the Swiss electricity mix and are modeled with the cut-off approach. The comparison is done for those impact categories that have been considered in both studies. Since the LCIA in the original study by Althaus and Gauch was calculated based on the older version of the Ecoinvent database v2.0, the impact assessment has been recalculated with background data from the current Ecoinvent version v2.2 in order to make the results comparable to this study. The resulting difference in total impacts between these three calculations is shown in **Table 14** for several indicators. Large differences in the original and new results of Althaus & Gauch are observed in the assessment with EI 99 human health. They are mainly due to the inclusion of heavy metal emissions from landfilling of sulfidic tailings in the new Ecoinvent version.

Table 14 Comparison of impacts between vehicle model by Althaus & Gauch and BEV baseline scenario

	BEV Althaus & Gauch Original (Ecoinvent v2.0)	BEV Althaus & Gauch New (Ecoinvent v2.2)	BEV Compact BL (Ecoinvent v2.2)
GWP	1.04E-01	1.06E-01	9.34E-02
CED Nuclear	1.74E+00	1.79E+00	1.74E+00
CED Fossil	1.50E+00	1.53E+00	1.39E+00
CExD Metals	1.84E-01	1.64E-01	1.28E-01
EI 99 Human Health	1.51E-07	3.85E-07	3.18E-07
EI 99 Ecosystem Quality	2.00E-02	2.40E-02	2.15E-02
EI 99 Resource Quality	1.66E-01	1.70E-01	1.52E-01
EI 99 (H/A)	9.44E-03	1.59E-02	1.36E-02

The results of the BEV baseline model of this study are now compared to those of the passenger car model from Althaus and Gauch in the following figures. The comparison is only meaningful between this study's results and the recalculated results from Althaus and Gauch because they are based on the same background data. Hence, the discussion of the figures will be based on this comparison. The original results of Althaus and Gauch are however indicated to show the difference between the calculations with the older and current Ecoinvent version. The major difference between the calculations with the older and the current Ecoinvent version occurs in the assessment of human health impacts and the total EI score (see **Figure 17**). The impacts per impact category are presented on a relative scale, by setting the score of the electric compact car baseline scenario from this study to 100%.

Figure 17 shows the total impacts of both vehicle models per impact category. Comparing the BEV baseline model with the newly calculated impacts of the electric passenger car from Althaus and Gauch, it can be seen that the model from Althaus and Gauch results in 5%-30% higher impacts. The difference between the two models is the lowest for the nuclear energy demand and highest for the exergy demand for metals.

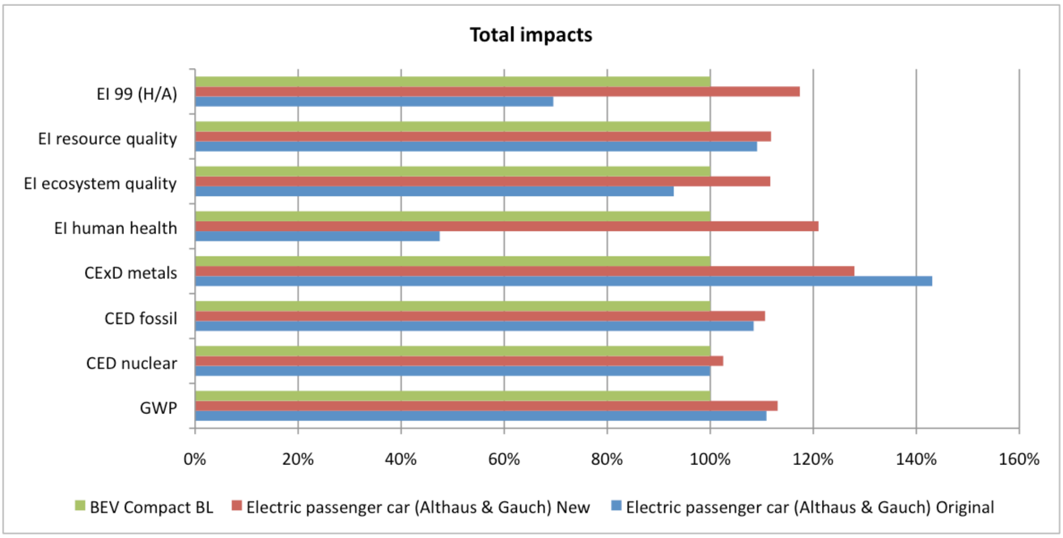

Figure 17 Comparison of total impacts between electric vehicle models

To analyze which processes are causing this difference in total impacts, the glider and drivetrain, the battery and the maintenance processes of both vehicle models are compared separately in Figure 18 to Figure 20. The impact scores have been calculated on a relative scale and are compared for each indicator. Processes concerning the road as well as the disposal of the vehicle have been modeled equally and therefore result in equal impacts if calculated with the same Ecoinvent background data. Hence, they are not presented here. The impacts from energy consumption and emissions in the use phase of the car are very similar for both models assessed with current Ecoinvent data. They are only 0.3% lower for the baseline model. Therefore, they are not presented in a separate graph either.

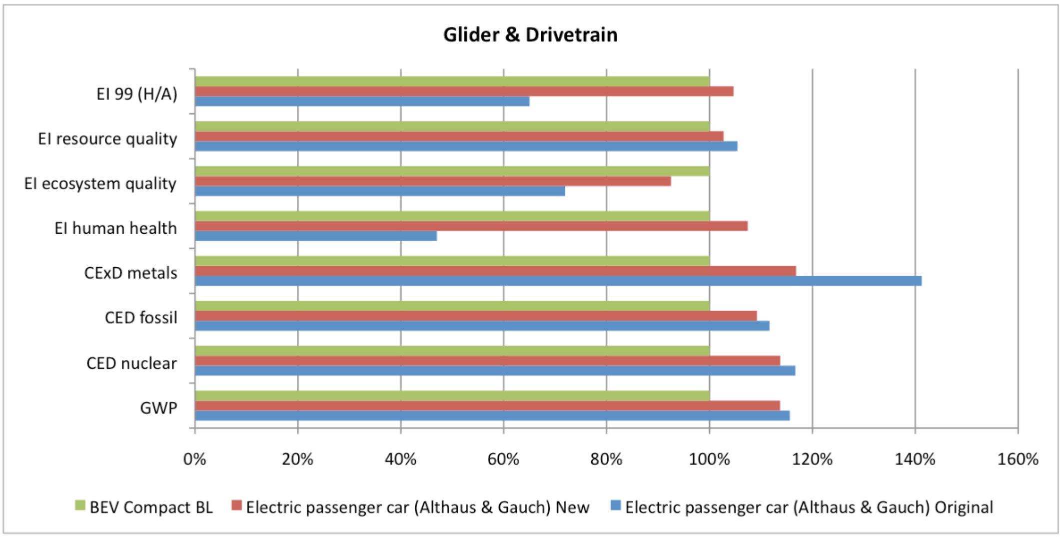

Figure 18 Comparison of impacts from glider and drivetrain between electric vehicle models

Impacts of the glider and the drivetrain of the electric car model from Althaus and Gauch are 5%-15% higher for all indicators except for ecosystem quality, as shown in Figure 18. Impacts on ecosystem quality are 7% higher in the baseline compact car model.

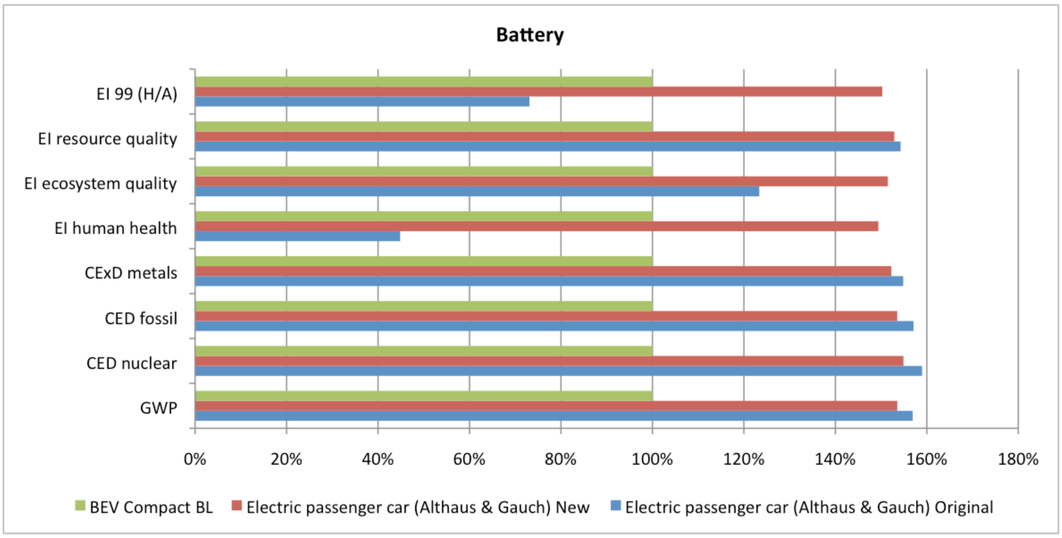

Figure 19 Comparison of impacts of the battery between electric vehicle models

In the comparison of impacts from the battery presented in Figure 19, the battery of the electric car model from Althaus and Gauch has roughly 50% higher impacts than the BEV model in this study. Since the only difference is the weight of the battery, this is not very surprising however. Both vehicle models apply the same data to model the battery lifecycle.

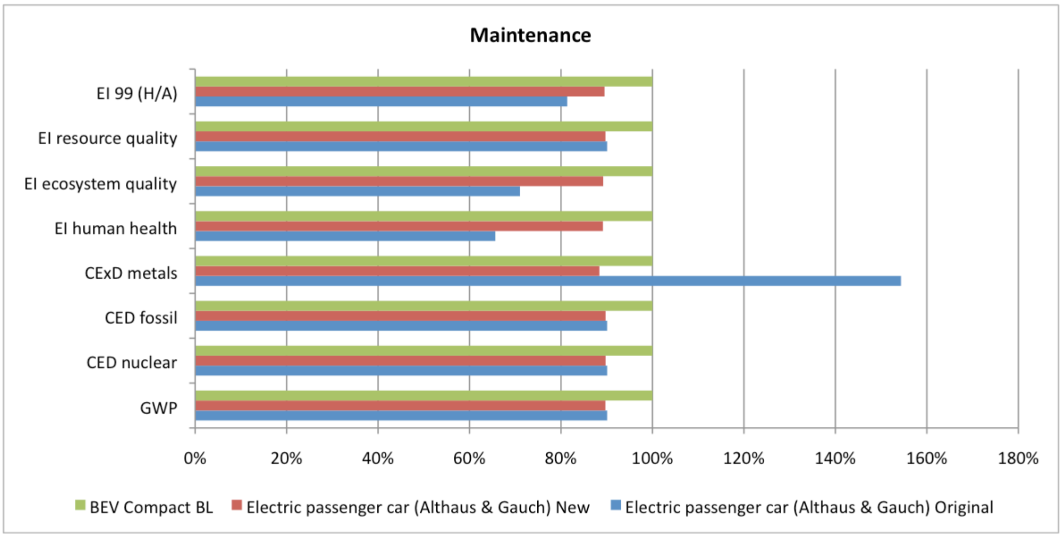

Figure 20 Comparison of impacts from maintenance between electric vehicle models

Impacts from maintenance are 10%-13% higher for the compact car baseline model with respect to all impact categories. The difference in the modeling approach of maintenance is that Althaus and Gauch have applied a maintenance process without considering the vehicle weight. In this study, the maintenance process has been scaled according to the vehicle weight in all vehicle scenarios.

7 SENSITIVITY ANALYSIS

The results of the LCIA depend to a certain degree on assumptions and methodological definitions concerning the model of the product life cycle. Therefore, a sensitivity analysis is performed in order to analyze the influence of such assumptions on the results of the LCA. The two aspects of the vehicle model that are assumed to have a critical influence are the electricity supply in the use phase of BEV and the modeling approach of EOL processes.

Since the study was conducted from a Swiss perspective, the Swiss electricity mix was applied to power the electric vehicles in the use phase in the basic model. However, the Swiss electricity mix consists of a high share of hydropower and nuclear power, having relatively low impacts on global warming. If the results of this study should be translated to a European or international context, the Swiss electricity mix would not be representative. Therefore, it is interesting to analyze the difference in impacts when the car is powered by electricity from a different country mix with a higher share of fossil electricity sources. Hence, the average European mix is applied to power the vehicle in order to recalculate the impacts and to test the sensitivity of the LCIA results depending on the choice of electricity mix.

In the basic model of the product life cycle, end-of-life processes were modeled with the cut-off approach. Since recycling and waste treatment processes are neglected in the cut-off model, the product life cycle was modeled in a second version with system expansion and substitution (see 5.3). The LCIA was then calculated again in order to test the influence of the allocation approach on the results.

7.1 Electricity Supply

The impact of the electricity mix on the LCIA results is tested by recalculating the results presented in chapter 6.2 with the Ecoinvent dataset of the UCTE production mix instead of the Swiss mix to power the electric vehicle in the use phase. UCTE stands for *Union for the Coordination of Transmission of Electricity* and the UCTE mix represents the average electricity supply mix in Europe. The analysis of the sensitivity of the results in dependence of the electricity mix is presented only for the baseline compact car vehicles since the effect is the same for all BEV scenarios. In **Figure 21**, the use phase impacts of the three vehicle scenarios compact car ICE, compact BEV Swiss mix and BEV UCTE mix are compared with all impact assessment methods. The Swiss mix scenario is defined as the reference case and the impacts of the other two options are scaled in relation to the reference for all indicators.

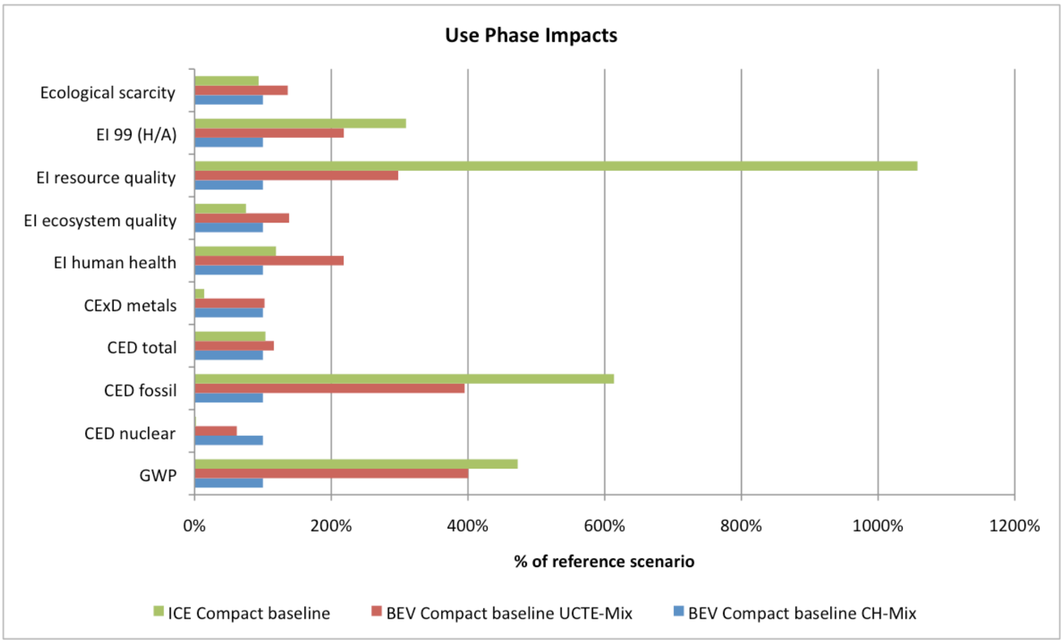

Figure 21 Sensitivity analysis of use phase impacts depending on the electricity mix

The influence of the electricity mix on the use phase impacts depends on the impact category. The UCTE mix leads to three times higher impacts than the Swiss mix concerning global warming and fossil energy demand. With respect to the indicators EI 99 (H/A), resource quality and human health the impacts are twice as high. The impact on ecological scarcity is slightly higher and the nuclear energy demand slightly lower if the UCTE mix is applied. The effect in the remaining impact categories is negligible. Including the ICE scenario in the comparison, it still causes considerably higher impacts than the electric vehicle powered by the UCTE mix in the categories GWP, fossil energy demand and EI 99 (H/A). Concerning impacts on resource quality, the ICE vehicle has much higher impacts than the electric, irrespective of the electricity mix. In the categories GWP, fossil energy demand and the total EI score however, the difference in impacts between the electric and the ICE vehicle becomes much smaller by calculating with the UCTE mix. The impacts on human health of the electric vehicle are lower than those of the ICE car if the Swiss mix is applied, but they become almost twice as high if calculated with the UCTE mix.

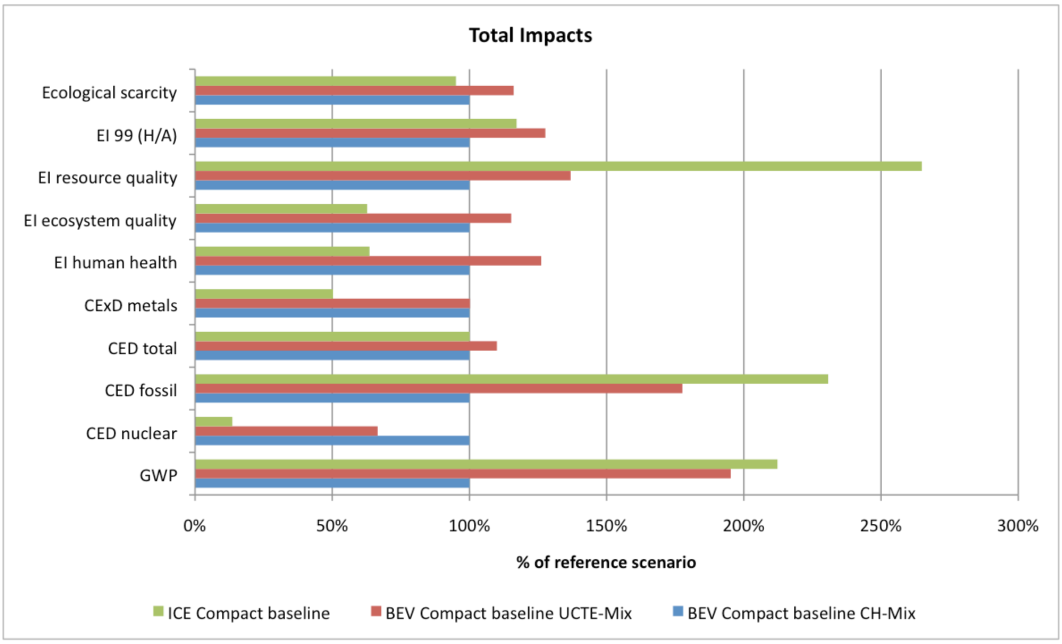

Figure 22 Sensitivity analysis of total impacts depending on the electricity mix

The sensitivity of the total life cycle impacts in dependence of the use phase electricity mix is presented in **Figure 22**. Again, the three compact car baseline options are compared and the impact scores are scaled in relation to the Swiss mix reference scenario in each impact category. Concerning the indicators EI 99 (H/A) and total energy demand, the impacts of the electric car become higher than those of the ICE car if the UCTE mix is applied instead of the Swiss mix. In contrast, the ICE car remains the option with the highest impacts with respect to EI resource quality, fossil energy demand and global warming potential. The UCTE mix scenario causes 15%-35% higher impacts than the Swiss mix scenario regarding the four ecoindicator categories as well as ecological scarcity. Concerning fossil energy demand and GWP the score is almost twice as high. The only impact category resulting in 30% lower impacts by applying the UCTE mix is the nuclear energy demand. The CExD for metals is equal in both electricity scenarios.

7.2 End-of-life and allocation

The impact assessment results of the EOL model with system expansion and substitution are first analyzed for three selected impact categories: GWP, CExD metals and EI 99 (H/A) (7.2.1-7.2.3). The results are presented in two graphs for each of the three assessment methods. In the first graph, the results are shown for all vehicle scenarios and the impacts are split between the main vehicle life cycle phases according to the definitions in **Table 15**.

Table 15 Declaration of labeling of impact assessment results (substitution model)

Labels	Processes
Road	Construction road + maintenance road + disposal road
Glider production	Production of glider
Drivetrain production	Production of drivetrain
Battery production	Production of battery + battery replacement
Energy & Emissions	Energy consumption + exhaust emissions + non-exhaust emissions
Maintenance car	Maintenance of car (without battery replacement)
Disposal glider	Recycling + waste treatment of glider materials
Disposal drivetrain	Recycling + waste treatment of drivetrain materials
Disposal battery	Recycling + waste treatment of battery materials (incl. replacement battery)

In the second graph, the impacts of the main life cycle phases modeled with the substitution approach are compared to those modeled with the cut-off approach. This comparison is only shown for the electric compact car baseline scenario because the other vehicle scenarios would show a similar pattern. The labels used to describe the life cycle phases in this second graph are defined in **Table 16**.

Table 16 Declaration of labeling of impact assessment results (comparison cut-off vs. substitution model)

Labels	Processes
Road	Construction road + maintenance road + disposal road
Glider	Production of glider
Drivetrain	Production of drivetrain
Battery	Production of battery + battery replacement
Energy & Emissions	Energy consumption + exhaust emissions + non-exhaust emissions
Maintenance car	Maintenance of car (without battery replacement)
Disposal car	Recycling + waste treatment + disposal of glider, drivetrain and battery

In the second part of this analysis, the effect of system expansion and substitution is compared to the cut-off model for several impact assessment methods, in order to assess the sensitivity of the LCA results depending on the EOL modeling approach. The comparison is shown for the electric compact car vehicles in the baseline, the lightweight plastic and the lightweight aluminum scenario. Further, the compact car baseline scenario of the electric and the ICE vehicle is compared in the substitution model. With regard to the smaller sized vehicles, the overall effect would be the same. Hence, these results are not included in the figures for reasons of readability.

In the figures presenting the substitution model results, some impact scores are negative. These negative scores have to be interpreted as avoided impacts. Negative impact scores arise from the disposal of the glider, the drivetrain and the battery, because these components are subject to recycling and waste treatment processes at their EOL. Due to material recovery from recycling of metals, the use of primary materials can be diminished, which is expressed by the negative impact scores.

Further, the waste treatment processes of synthetic materials generate heat and electricity as by-products, which can in turn substitute energy production from primary resources.

7.2.1 Global warming potential

Life cycle impacts on global warming of the 12 vehicle scenarios have been assessed with the substitution model in kilograms of CO_2-equivalents per vehicle kilometer. The results are presented in Figure 23 by comparing the shares of the impacts of the main product life cycle phases.

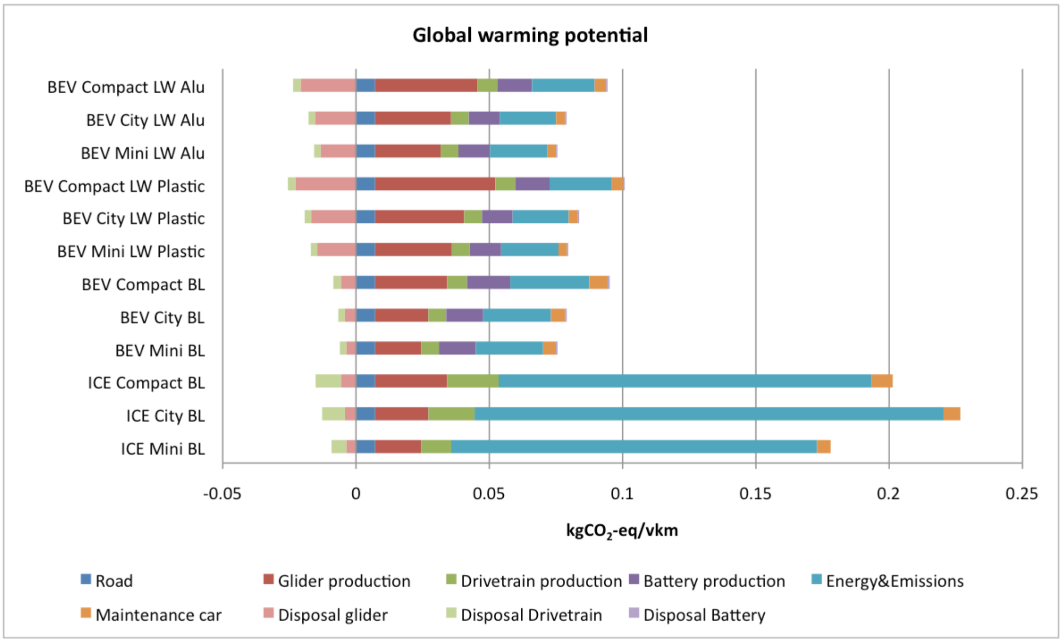

Figure 23 GWP of vehicle scenarios modeled with system expansion and substitution

In the electric vehicle scenarios, the main effect of EOL processes stems from the disposal of the glider. This is mainly due to the effect of material recycling, which is biggest for the lightweight plastic scenario and smallest for the baseline scenario. In both lightweight scenarios, the glider disposal processes compensate roughly half of the impacts from the production of the glider. Recycling processes of the electric drivetrain do not have a significant contribution. In contrast, EOL processes concerning the ICE drivetrain have a bigger effect compared to the electric drivetrain. However, in view of the total impacts of the ICE scenarios the share of impacts avoided in the disposal phase is marginal.

Figure 24 presents the comparison of impacts on global warming between the electric compact car baseline scenario modeled with the cut-off and the substitution approach.

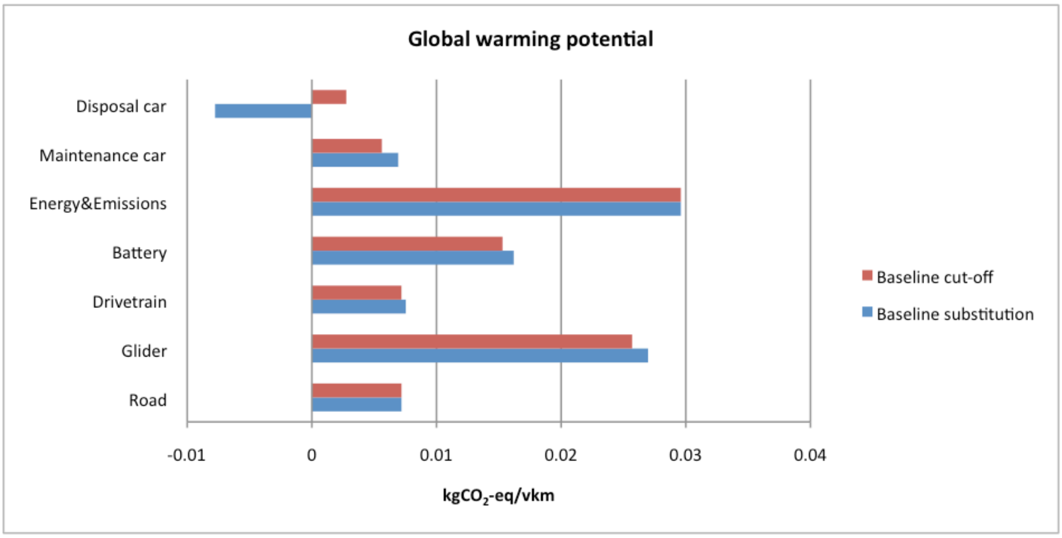

Figure 24 Comparison of GWP between cut-off and substitution model

The impacts on global warming from the disposal of the car are positive in the cut-off model, but they are negative in the substitution model. The production and maintenance processes of the car have slightly higher impacts if modeled with substitution, however. Impacts from the use phase are the same in both modeling approaches because they are modeled equally.

7.2.2 Cumulative exergy demand metals

The results of the LCIA modeled with system expansion and substitution are presented with respect to the cumulative exergy demand for metals in Figure 25. The impact scores are calculated in mega joules equivalents per vehicle kilometer.

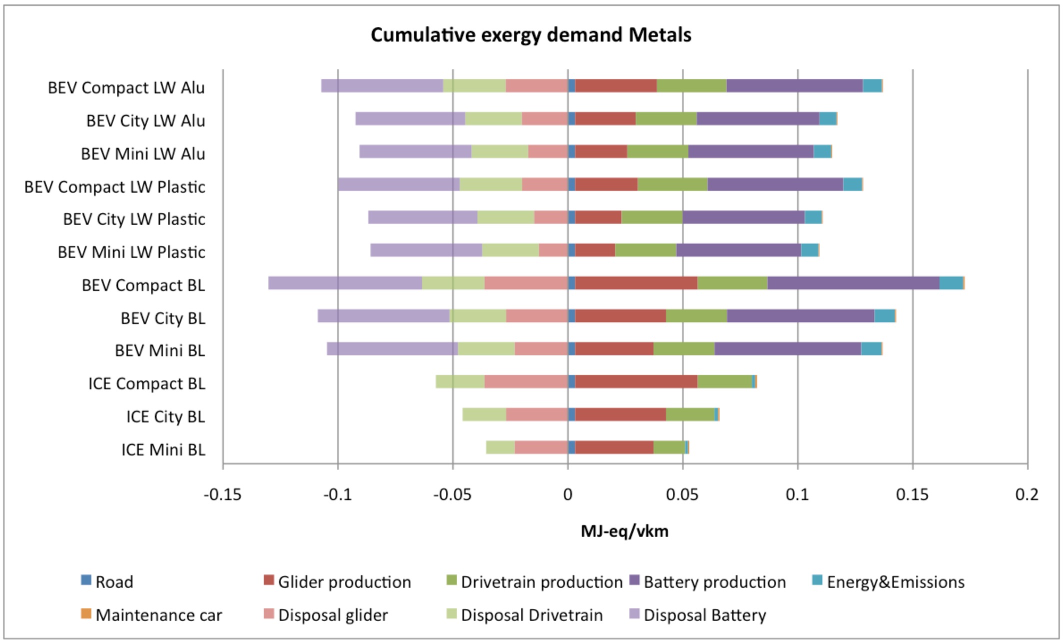

Figure 25 CExD metals of vehicle scenarios modeled with system expansion and substitution

EOL processes have a considerable effect on the exergy demand for metals. Recycling and waste treatment processes can substitute about two-thirds of the demand from production and use of the vehicle. Concerning the electric vehicles, the battery has the highest contribution, while regarding the ICE vehicles the glider contributes the most to the reduction potential of the disposal processes.

In **Figure 26** the impacts on the exergy demand for metals are compared between the electric compact car scenarios modeled with the cut-off and the substitution approach.

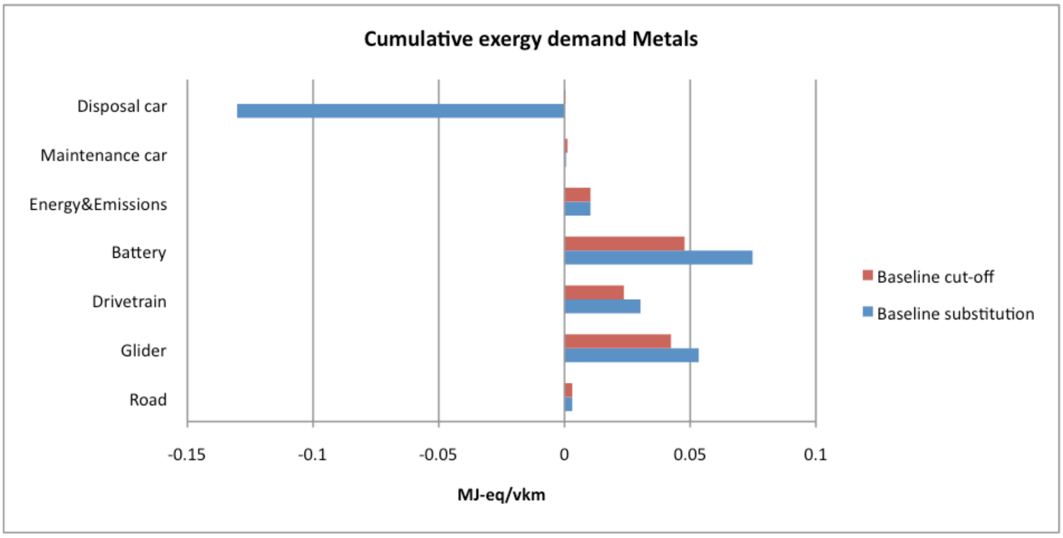

Figure 26 Comparison of CExD metals between cut-off and substitution model

Comparing the cut-off and the substitution model, the disposal of the car has a strongly negative score in the substitution model, whereas in the cut-off model the contribution of the disposal process is negligible. Concerning the battery production, the substitution model results in 50% higher impacts. The production of the glider and the drivetrain also result in slightly higher impacts if modeled with substitution.

7.2.3 Ecoindicator single score

Life cycle impacts of the vehicle scenarios modeled with system expansion and substitution have further been assessed with ecoindicator 99. The results are presented in Figure 27.

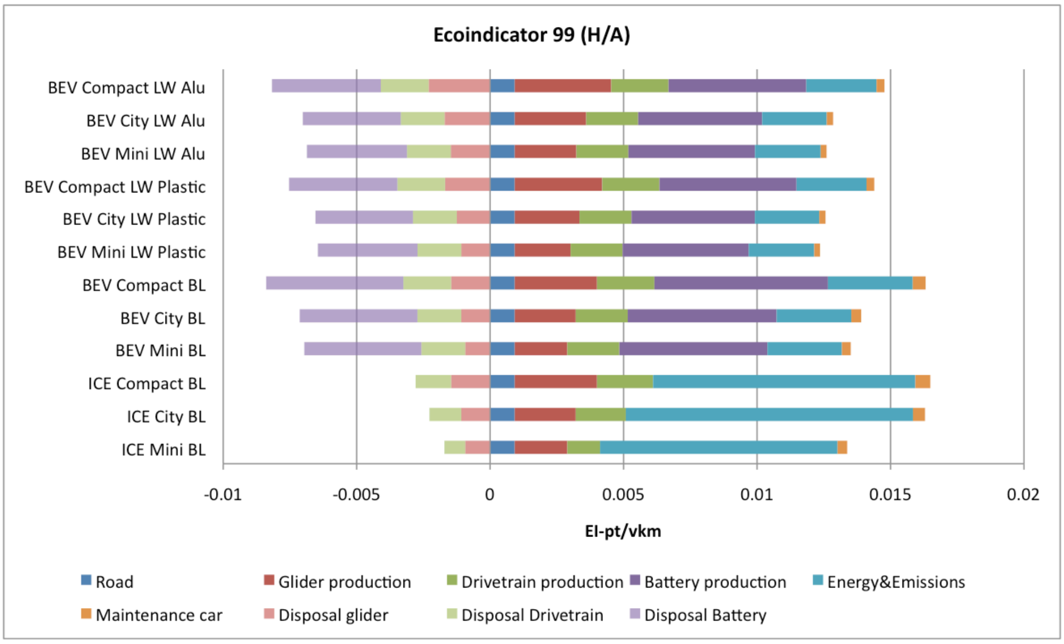

Figure 27 EI scores of vehicle scenarios modeled with system expansion and substitution

The disposal processes of the electric vehicle scenarios reduce roughly half of the production and use phase impacts measured with ecoindicator. The highest contribution stems from the disposal of the storage battery. For the ICE vehicles however, the effect of EOL processes is quite small compared to the total impact score.

In Figure 28 the impacts of the electric compact car scenarios assessed with ecoindicator are compared between the cut-off and the substitution model.

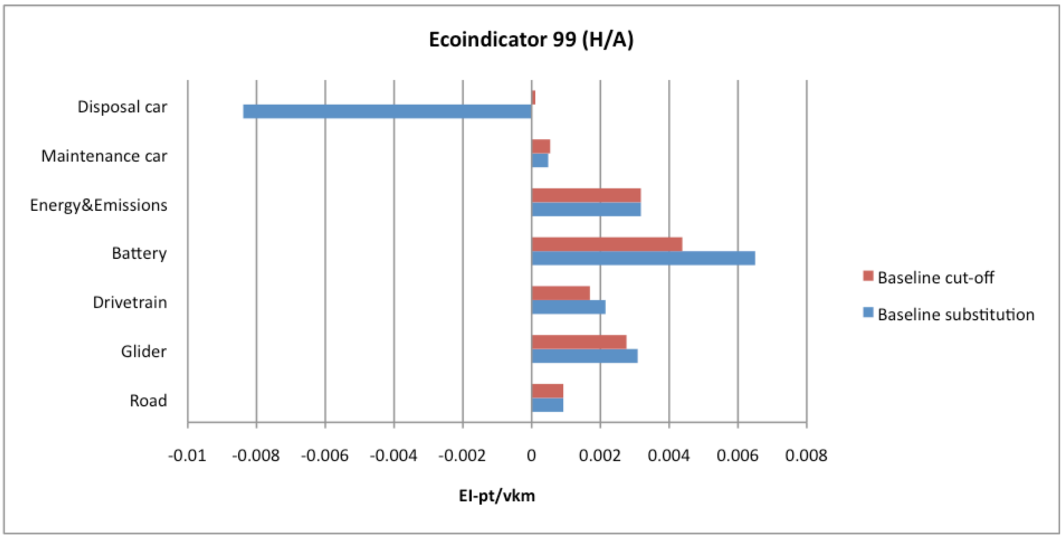

Figure 28 Comparison of EI scores between cut-off and substitution model

In the cut-off model, the disposal of the car has a very small impact, whereas in the substitution model, EOL processes have a high impact reduction potential. Impacts from the battery production are about 50% higher if modeled with system expansion and substitution. The production of the glider and the drivetrain also results in slightly higher impacts in the substitution model.

7.2.4 Comparison of cut-off and substitution model results

The results of the impact assessment of the substitution model are now compared to those of the cut-off model with respect to several impact categories. The comparison is shown for the electric compact car vehicles in the baseline, the lightweight plastic and the lightweight aluminum scenario in Figure 29. The baseline vehicle is chosen as the reference scenario and the impacts of the other vehicle scenarios are scaled in relation to the reference case for all indicators.

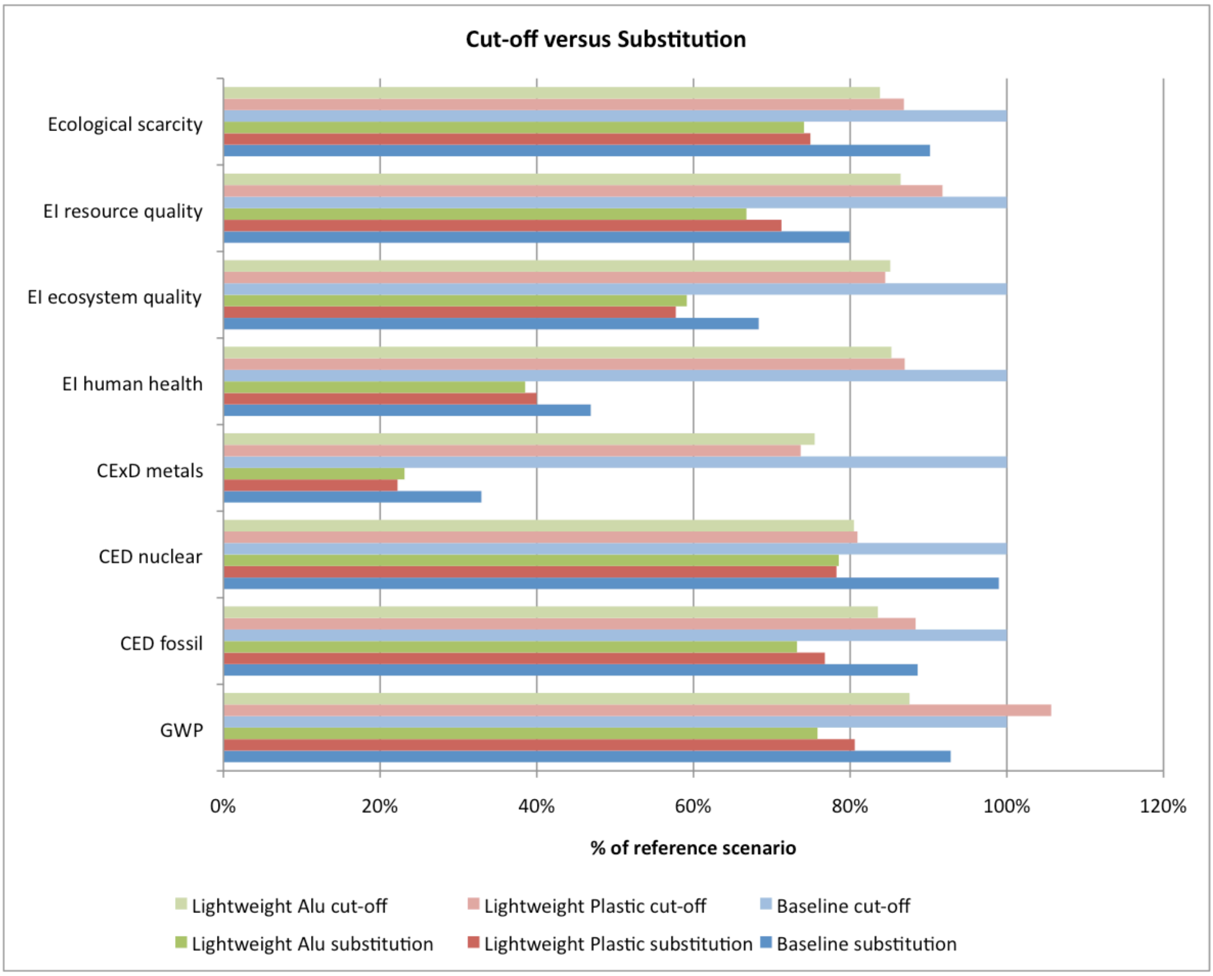

Figure 29 Comparison of LCIA results between cut-off and substitution model

The total life cycle impacts of the electric car are lower if modeled with system expansion and substitution compared to the cut-off model with respect to all impact categories. The order of best-case and worst-case scenario remains the same, whether modeled with cut-off or substitution, for all indicators except for the GWP and the nuclear energy demand. In the cut-off model, the lightweight plastic scenario has the highest impacts on global warming, whereas in the substitution model the baseline scenario causes the highest impacts, as in all other impact categories. Concerning nuclear energy, the lightweight plastic has higher impacts than the lightweight aluminum scenario in the cut-off model, whereas in the substitution model it is the other way round. However, the difference between the two scenarios is very small. The variation in impacts between the two modeling approaches depends on the impact category. It is biggest concerning the CExD for metals and smallest for the nuclear energy demand. Impacts are about half to one-third smaller if modeled with substitution for the indicators CExD metals, EI human health and ecosystem quality. For the other indicators, except CED nuclear, total impacts are 10%-20% lower if modeled with substitution. Concerning the nuclear energy demand, there is almost no difference in impacts between the two models.

The effect of the substitution model is now assessed with respect to the comparison between the electric and the conventional vehicle options. In **Figure 30**, the impacts of the compact car baseline scenario of the electric and the ICE car are compared in the substitution and the cut-off model. Again, impacts are calculated in relation to the electric vehicle modeled with the cut-off for all indicators.

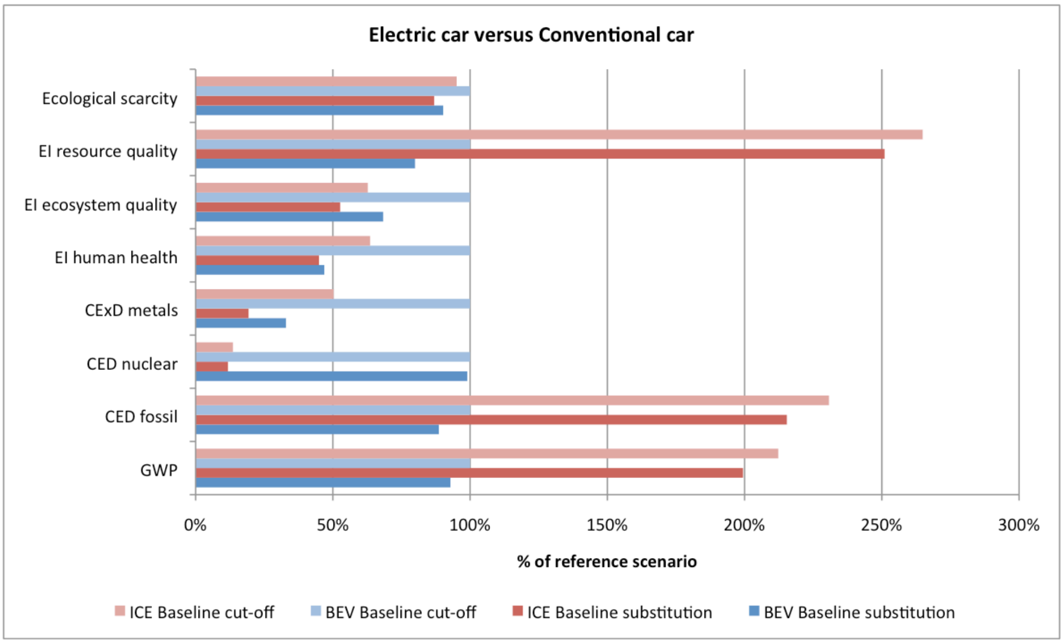

Figure 30 Comparison of electric and ICE vehicle with cut-off and substitution model

Both the electric and the ICE vehicle result in lower life cycle impacts if modeled with system expansion and substitution in all impact categories. Further, the order of environmental performance does not differ between the cut-off and the substitution model. However, the effect of the change in the modeling approach is not the same for all indicators. Regarding those impact categories that result in much higher impacts for the ICE vehicle, the difference in impacts between the electric and the ICE car is similar in both models. This is the case for the fossil energy demand, the GWP and resource quality. However, with respect to those indicators assessing higher impacts for the electric car, the difference between the cut-off and the substitution model results is much bigger for the electric car. Therefore, the difference in impacts between the ICE and the electric vehicle becomes smaller in the substitution model concerning ecosystem quality, human health impacts and exergy demand for metals. A special case is the assessment of the nuclear energy demand, which shows almost the same results in both modeling approaches.

8 INTERPRETATION

8.1 Impact assessment results

The question whether, from an environmental point of view, electric vehicles are preferable to conventional ICE vehicles has to be answered in two different lines of argumentation. One is focused on global warming and fossil resources, the other on toxic effects and damages to ecosystems. In the comparison of life cycle impacts between electric and ICE cars, the two vehicle options show significantly different results in the assessment with eight environmental damage indicators. The ICE vehicle scenarios have shown clearly higher environmental impacts with respect to the impact categories ecoindicator resource quality, fossil energy demand and GWP. These indicators all refer to environmental impacts concerning global warming and the depletion of fossil resources. In contrast, the electric vehicle scenarios have resulted in higher total impacts with respect to the impact categories ecosystem quality and human health, exergy demand for metals and nuclear energy demand. Hence, the major environmental impacts of electric vehicles are toxic effects on human health and ecosystems as well as the depletion of metallic resources. It has to be considered though that the assessment was based on the Swiss situation with power supply in the vehicle use phase from an electricity mix low in GHG emissions. Therefore, climate change impacts in the use phase of the electric vehicle scenarios are very low, whereas the demand for nuclear energy sources is very high. Nevertheless, it can be argued that depending on which categories of impacts are weighted stronger, either the ICE or the electric vehicle causes higher impacts to the environment.

In the BEV scenarios, the storage battery is responsible for a large share of impacts in those categories, which result in higher total impacts compared to the ICE vehicle scenarios. The assessment with ecoindicator showed that the battery production mainly causes human health impacts and the degradation of ecosystems. In line with that, the battery's impacts on ecological scarcity are primarily caused by emissions into the air and the surface water, as well as by waste deposits. Analyzing these impacts in more detail, it becomes clear that the copper used in the anode material contributes the most to the impacts on human health and ecosystems as well as to the emissions into the air. Emissions into the surface water stem mainly from mining and processing of gold, which is used for the production of the electronic components. Waste deposits contributing to ecological scarcity originate partly from electronic components, partly from radioactive waste material from nuclear energy for the processing of electronics. Copper and gold also have the highest contribution to the exergy demand for metals in the battery production.

If only the two drivetrain options (without the battery) are compared, it depends on the assessment method, whether the electric or the ICE drivetrain causes the higher impacts. The electric drivetrain has higher impacts on human health and the exergy demand for metals. The ICE drivetrain in contrast has a higher energy demand and GWP, as well as a higher impact on ecological scarcity. It has to be considered though that the ICE drivetrain is much heavier than the electric (without the storage battery). Therefore, higher impacts of the ICE drivetrain are caused because of energy efforts for its production and the extraction and processing of metals, mainly steel and aluminum. These impacts refer to global warming and the energy demand. Concerning ecological scarcity, the ICE drivetrain primarily causes emissions into the air and the surface water. These emissions stem to a very high degree from polyphenylene sulfide. Further, platinum has a significant contribution to the emissions into the surface water. The electric drivetrain in contrast has a higher share of electronic components, which are responsible for human health impacts and the exergy demand for metals. The most important metals are copper, aluminum, silver and gold. Copper is used in the windings of the electric motor, as well as in electronic components. Aluminum, silver and gold are as well applied in the production of electronics. The impacts on human health from the electric drivetrain stem mainly from the same four metals. In contrast, human health impacts of the ICE drivetrain are primarily caused by platinum and aluminum. The conclusion regarding the impacts from the drivetrain is in line with the comparison of total life cycle impacts between the electric and the ICE vehicle scenarios, suggesting that depending on the weighting of environmental impacts one or the other option is preferable.

In the comparison of the BEV baseline scenarios the effect of the vehicle size has been shown. If the vehicle size is reduced from a golf class to a mini class vehicle, environmental impacts are 15%-20% lower in the baseline scenario. The effect is similar regarding all impact categories. This can be explained by the fact that a reduced vehicle size affects all life cycle processes. If the vehicle is smaller, less material input and production efforts are needed for the manufacturing of the glider. A small vehicle needs a less powerful motor and the storage capacity of the battery can be lower as well. Hence, the impacts from the drivetrain and the battery are also reduced. If the vehicle is lighter, the demand for propulsion energy is smaller (allowing a lighter storage battery) and use phase impacts are reduced. Therefore, size-reduction of the vehicle has a positive effect on the whole environmental performance.

In the assessment of the lightweight scenarios, two different effects have to be differentiated. The one is the effect of the overall vehicle weight reduction because of the reduced glider and battery weight. This argumentation goes in line with the explanations on the effect of size-reduction. The other effect concerns the change of material composition in the glider. The overall weight reduction has a positive effect on total life cycle impacts of a 10%-25% reduction (depending on the vehicle scenario) regarding all impact categories, except GWP. Concerning global warming, the lightweight plastic vehicles have higher total impacts than the baseline vehicles even though their overall weight is considerably lower, corresponding to lower use phase and battery production impacts. This means that the production of a glider based on synthetic materials causes considerably higher impacts on global warming compared to the baseline. A more detailed analysis of the GWP of the lightweight plastic glider leads to the finding that the largest impacts are caused by magnesium, followed by electricity inputs, aluminum and epoxy resin production. In the baseline and the lightweight aluminum scenarios, the major contributions to the GWP stem from steel in the baseline and aluminum production in the lightweight aluminum scenario. In the assessment with ecoindicator, the baseline scenario has the highest impacts, which mainly stem from the production of steel and electronic components. In the lightweight plastic glider, the aluminum production and the use of epoxy resins have the highest contribution, whereas in the lightweight aluminum glider copper and aluminum contribute the most. Concerning the exergy demand for metals, the high amount of steel in the baseline glider is responsible for roughly half of the impacts, whereas in both lightweight scenarios electronic components, copper and aluminum are causing the main impacts. Generally, the lightweight aluminum scenario results in lower impacts in all categories than the lightweight plastic scenario. The reason becomes clear if the impacts of the glider production are compared per kilogram of glider material. The materialization of the glider based on synthetics leads to the highest impacts per kilogram whereas the baseline composition has the lowest impacts in all categories (except the exergy demand for metals, which is very similar in all scenarios). Concerning the GWP, the impacts of the plastic-based glider are three times higher than the baseline per kilogram of material. In the whole life cycle, the higher impacts of the glider material outweigh the effects of the weight reduction in the lightweight plastic scenario with respect to global warming. Contrary, in the assessment of all other impact categories lightweight construction based on synthetics leads to lower life cycle impacts. This means that the effect of reduced impacts in the other life cycle phases outweighs the higher impacts of the production of the lightweight plastic glider. The production of the aluminum-based glider also leads to higher impacts per kilogram compared to the baseline in all impact categories (except CExD metals). However, the positive effect of the reduced vehicle weight compensates the higher glider production impacts, resulting in considerably lower life cycle impacts of the lightweight aluminum scenario compared to the baseline scenario. Hence, it can be concluded that lightweight construction is effective in reducing overall impacts of electric vehicles. However, the effect depends on the choice of lightweight materials for the substitution of iron and steel.

The LCA results of the electric compact car baseline scenario have been compared to the impact assessment results of the golf class EV model by Althaus and Gauch (Althaus and Gauch 2010). It was found that their vehicle model results in higher total environmental impacts with respect to all impact categories. The difference in impacts between the two vehicle models is due to the production and maintenance processes, whereas the use phase impacts are very similar. The battery leads to about 50% higher impacts in the study by Althaus and Gauch compared to the BEV baseline scenario. The reason is that the battery applied in their study is much heavier with a weight of 400 kg compared to the battery weight of 260 kg in the compact car baseline scenario. The dataset used to model the battery life cycle is however the same in both studies. Differences in impacts between the compared EV models are further due to the different glider and electric drivetrain models. The glider of the baseline scenario results in higher impacts compared to the glider model by Althaus and Gauch in all categories. The cause is primarily the higher amount of reinforcing steel used in the glider production. In contrast, the model of the electric drivetrain by Althaus and Gauch leads to higher impacts compared to the electric drivetrain modeled in this study with respect to all indicators, except resource quality. The main reason is the higher amount of electronic components. Althaus and Gauch have inventoried 4 kg of electronic components, whereas in this study the amount of electronics in the electric drivetrain is assumed to be only 1.6 kg. The use of copper, aluminum, silver and gold in the electronic components is the main reason for the impacts on human health, resource quality and the exergy demand for metals. Further, the amount of aluminum is also higher in the drivetrain model by Althaus and Gauch, contributing to the higher impacts on resource quality and global warming. In contrast, the drivetrain model in this study contains a higher amount of copper and chromium steel, mainly contained in the electric motor, causing the higher impacts on ecosystem quality. In summary, the higher impacts of the electric drivetrain outweigh the lower impacts of the glider in the model by Althaus and Gauch, except for impacts on ecosystem quality that are higher for both glider and drivetrain in this study's vehicle model. There is further a slight difference in the impacts from maintenance in the compared vehicle models, which are slightly higher in this study. The reason is that a higher amount of maintenance materials is inventoried, adjusted to the vehicle weight. The impacts from maintenance are caused mainly by the use of aluminum, steel and copper and the electricity consumption.

It has been shown that the use of copper is responsible for a large share of impacts, especially impacts on human health and ecosystem quality. This can also be seen from the comparison of the calculation of the vehicle model by Althaus and Gauch with older and current Ecoinvent data. Total impacts on human health of the EV are almost three times higher if calculated with the new Ecoinvent version. The main reason is the model of the mining of copper. In the previous version of the Ecoinvent database, heavy metal emissions from the disposal of sulfidic tailings arising during copper production have not been included. In the current version however, such emissions are accounted for, resulting in much higher impacts (especially on human health) for the whole copper production process.

8.2 Sensitivity analysis of electricity supply

The sensitivity analysis of the electricity supply has shown that the LCIA results of the vehicle scenarios are influenced by the choice of electricity mix applied in the vehicle use phase. The electric vehicles have higher total impacts if powered with the UCTE mix compared to the Swiss mix in all impact categories, except for the nuclear energy demand. If the use phase impacts are calculated with the UCTE mix, the demand of fossil resources and the GWP are twice to three times higher. Therefore, it becomes clear that e-mobility is only a valid strategy for climate change mitigation if the vehicles are powered with electricity from sources low in GHG emissions. Compared to conventional ICE vehicles, the electric vehicles still have slightly lower impacts on global warming and the depletion of fossil resources even if they‘re powered by electricity from the UCTE mix. Regarding impacts on resource quality, the ICE vehicle has considerably higher impacts, irrespective of the power supply for the BEV. It can be concluded that the type of electricity mix does not change the order of environmental performance in the comparison of ICE with electric cars. However, the advantage of the BEV regarding its climate change mitigation potential is reduced if the electricity is produced from carbon-rich sources. Therefore, it becomes clear that the Swiss situation is very favorable for electric mobility in the light of the climate debate, since the national electricity mix consists mainly of hydropower and nuclear energy. The question whether nuclear power should be considered a sustainable option to substitute fossil energy sources shall not be further discussed here. However, the disadvantages of this technology are accounted for if an aggregating indicator such as ecological scarcity is considered for the assessment. Further, it has to be mentioned that the UCTE mix is only an average mix and that specific conclusions for other European countries should be based on more detailed analyses of the respective situation. The sensitivity analysis of the electricity supply leads to the general conclusion that e-mobility is a potential strategy for climate change mitigation in countries with an electricity mix low in GHG emissions. However, if the electricity supply depends to a high degree on fossil resources, the effect of electric vehicles on climate change abatement is diminished.

8.3 Sensitivity analysis of end-of-life modeling approach

The comparison of the impact assessment of the substitution model with the cut-off model shows that EOL processes of the glider, the drivetrain and the battery have an important influence on the results. Avoided impacts in the substitution model arise due to two different effects. The one is the substitution of primary materials from recycling of metals; the other is the production of energy because of waste treatment processes. In the evaluation of the results of the substitution model, three impact categories have been analyzed in more detail: global warming, exergy demand for metals and the assessment with ecoindicator. Regarding the exergy demand for metals, avoided impacts in the substitution model can only stem from recycling processes, because there is no energy production from metallic wastes. The total GWP in the substitution model can be lowered because of energy production from waste treatment substituting energy production from fossil sources, but mainly because of recycling processes avoiding the impacts from primary metal production. Similarly, in the assessment with ecoindicator both types of EOL processes can substitute for a share of the primary production and result in lower total impacts. The effect of substituting primary materials for recycled secondary materials is much more important however, as will be further discussed. In order to evaluate which materials are causing the reduction of impacts in the substitution model compared to the cut-off, the EOL of the main vehicle components has to be analyzed separately.

EOL processes of the battery only reduce the scores of the exergy demand for metals and the ecoindicator, but not the GWP. It has been shown that copper has a large impact on the LCA results of the battery. From the substitution model, it can be concluded that recycling of copper, which of a high amount is contained in the anode material, can avoid a large part of the impacts of the battery. The substitution of primary copper diminishes the impacts from the disposal of sulfidic tailings from copper mining, which is the main cause for the high impacts assessed with ecoindicator. Concerning global warming, the main impacts from the production of the battery stem from primary aluminum and lithium in the cathode. The GWP of the disposal of the battery however has a positive score. Recycling can only avoid a very small share of impacts, but the disposal of electronics – mainly the disposal of lead, nickel and gold components – increase the battery's impacts on global warming.

In the substitution model, the EOL of both drivetrain options mainly influences the exergy demand for metals and the ecoindicator score, but only has a very small impact on the GWP. The influence of recycling on the CExD metals seems to be rather evident, since the drivetrain contains a high amount of aluminum, steel and copper. Primary copper production is responsible for the main impacts of the electric drivetrain on the CExD metals. Hence, the substitution thereof due to copper recycling can avoid a large share of impacts in the EOL phase. However, the second most important material is gold, contained in the electronic components of the electric drivetrain. The substitution of primary gold due to recycling can avoid a large share of the exergy demand for metals. The same conclusions as for the CExD metals hold true for the assessment of the electric drivetrain with ecoindicator. Avoided impacts in the assessment of the cumulative exergy demand and ecoindicator are of the same size for both the electric and the ICE drivetrain. It has to be considered though that the ICE drivetrain is much heavier and contains a larger amount of steel and aluminum. Therefore, the substitution effect for the ICE drivetrain is mainly due to recycling of steel and aluminum, whereas for the electric drivetrain recycling of gold from the electronic components and copper from the electric motor has a major impact. The effect of EOL processes on the GWP is small for both drivetrain options, but the disposal of the ICE drivetrain has a higher impact reduction potential than the electric drivetrain. The reason is that the GWP can primarily be reduced by the substitution of primary aluminum and steel for secondary materials because the production thereof is very energy intensive. Since the ICE drivetrain has a higher amount of steel and aluminum the effect of recycling is bigger compared to the electric drivetrain.

EOL processes of the glider also have a significant contribution to the difference in life cycle impacts between the substitution and the cut-off model. The impact category affected by the glider disposal however depends on the glider scenario. Concerning the lightweight scenarios, the main effect of the substitution model can be seen for the impacts on global warming. The main impacts on the GWP of the lightweight glider stems from the production of primary aluminum. Correspondingly, the highest share of global warming impacts can be avoided by aluminum recycling. Further, for the lightweight plastic glider the substitution of primary magnesium has a very high contribution to avoided impacts on global warming. Since the share of aluminum in the baseline glider is much lower than in the lightweight scenario, the effect of substitution on the GWP score is lower. In the baseline glider scenario, the biggest effect of the substitution model can be seen regarding the cumulative exergy demand for metals. The main contribution to the negative impact score results from the substitution of primary steel and copper as well as from recycling of gold used in electronic components. Concerning the ecoindicator score of the baseline glider, the same conclusion holds true, although the effect is smaller. For both lightweight scenarios, the cumulative exergy demand for metals is reduced due to the recycling of copper, aluminum and gold and the ecoindicator score is also mainly affected by the substitution of primary aluminum and copper.

The sensitivity analysis of the EOL model leads to the general conclusion that the LCIA results are different depending on the applied allocation approach. With all impact assessment methods, a substantial difference between the substitution and the cut-off model could be observed. The assessed impact scores of all vehicle scenarios are lower if modeled with system expansion and substitution for all indicators. However, the difference between the results of the two models depends on the impact category. The effect is smallest for the impacts on energy demand and global warming, and biggest for the impacts on ecosystem quality, human health and the exergy demand for metals. Further, the substitution model leads to a higher reduction of impacts for the electric vehicle scenarios compared to the ICE scenarios, because EOL processes of the battery have a major influence. Interestingly, the electric vehicles have higher impacts on human health, ecosystem quality and CExD metals than the ICE vehicles in the cut-off model. However, these are the impact categories resulting in the highest reduction of impacts if modeled with system expansion and substitution. Hence, in the substitution model the difference in impacts between the electric and the ICE vehicles regarding these indicators becomes much smaller. The main reason for this effect is the impact of the battery. In the cut-off model, the production of the battery causes high impacts on human health and ecosystem quality, leading to a worse environmental performance of the EV compared to the ICE car. In the substitution model however, these impacts can be reduced substantially, resulting in similar impacts of the electric and the conventional vehicle. This clearly shows the importance of recycling and waste treatment for the environmental performance of electric vehicles. Further, the results demonstrate the advantages of the system expansion compared to the cut-off approach. If the LCA is calculated only based on a cut-off model of the product inventory, the effect of waste treatment and recycling is neglected in the analysis. Hence, there is no incentive to foster such processes and it cannot be evaluated for which components of the product these processes would be most effective.

9 DISCUSSION

The research question of this study is asking about the main life cycle environmental impacts of electric passenger cars, and especially about the impacts from the vehicle glider and the electric drivetrain. From the LCIA it can be concluded that – in relation to the impacts from conventional vehicles – the main environmental impacts of electric cars are impacts on human health, damages to ecosystems and the depletion of metallic resources. The glider and the electric drivetrain contribute a considerable share to these impacts, ranging between one-third and half of the total life cycle impacts. It is however difficult to judge whether electric vehicles should be considered a more sustainable mobility option based on absolute impact scores. Therefore, the comparison with the assessment of conventional cars is important. In the assessment of the global warming potential and the depletion of fossil resources, the electric vehicles show a much better performance compared to conventional cars. This finding leads to the conclusion that e-mobility might have the potential to contribute to climate change mitigation. Nevertheless, the use of terms such as *zero-emission* or *no-pollution* in advertizing of electric vehicles is misleading or simply wrong, since environmental impacts do also arise for electric vehicles. Although they truly cause no tailpipe emissions, the use of fossil resources and emissions of GHG arise mainly during the vehicle production, but also from the power supply. In the use phase, the GWP of electric vehicles is much smaller compared to conventional cars, at least if assessed for the Swiss situation (or any other scenario powering the EV with electricity low in GHG emissions). However, if electric vehicles are powered with electricity from GHG-emitting sources, the impacts on global warming rise to an almost similar level as those of conventional vehicles. This has been shown in the sensitivity analysis of the power supply. The reason why the impacts of the electric car are still slightly lower if calculated with the European electricity mix is that the UCTE mix is not based 100% on fossil sources, but consists of a share of nuclear power and renewable energy sources. Generally, it can be concluded that the vehicle infrastructure has a higher weight in the assessment of the global warming potential of electric compared to conventional vehicles. Since use phase GHG emissions of EV are much lower, the impacts arising from the vehicle production have a higher importance. For the ICE vehicles in contrast, use phase GHG emissions are strongly dominating the life cycle impacts on global warming.

The differentiation between the main impacts of electric cars compared to the main impacts of conventional cars has further implications. It has been shown that e-mobility could be a potential strategy for climate change mitigation in a scenario comparable to the Swiss situation. And even if the power supply is based on a higher share of fossil resources such as the European electricity mix, electric vehicles are still preferable to conventional cars from the point of view of global warming. But the beneficial effects of e-mobility in terms of the climate change problem are mainly due to nuclear power in the assessment for the Swiss situation, because at present the Swiss electricity supply is strongly dependent on atomic energy. Therefore, the consequences of nuclear power generation have to be taken into account if electric vehicles are chosen as the preferable mobility option, under the assumption that the share of atomic energy in the electricity mix remains on a similar level in the near future. The disadvantages of nuclear power are, at least partly, accounted for if a fully aggregating indicator such as ecological scarcity is applied in the LCIA. At present, the share of renewable energy technologies (except for hydropower) in the total energy market is still very low in Switzerland. If electricity production from solar or wind power plants will grow substantially in the near future e-mobility might become even more preferable (it is assumed that hydropower has already reached a high share of its potential in Switzerland). This would only be the case however if new renewable energies were replacing a large share of today's nuclear power generation. The question arises what would be the consequences on the power supply if the market of electric vehicles is assumed to grow substantially in the future and if e-mobility should replace conventional vehicles in the long run. If electric vehicles are seen as the preferable mobility solution, the demand for electric power supply is certain to increase. It is however questionable if this surplus energy demand could be covered entirely by renewable energy technologies.

The impacts on human health and ecosystem quality have been shown to be most problematic for the environmental performance of electric vehicles in the cut-off model for the Swiss situation. And they result in even higher scores in the assessment with the European electricity mix. The reason is the increase in use phase impacts due to the electricity production, for example due to the release of NO_x, SO_x or PM_{10} particles, affecting the air quality and causing acidification and eutrophication of ecosystems. In the substitution model in contrast, human health impacts and ecosystem damages are much lower compared to the cut-off model, due to the effect of recycling and waste treatment, especially concerning the battery and the electric drivetrain. Concerning global warming, the disposal processes proved to have only a small effect on the reduction of the total GWP. It has to be considered though that there are some uncertainties in the model of the disposal processes. Since recycling schemes of battery and drivetrain materials from EOL electric vehicles are not yet well established, the model thereof is based on several assumptions. Further, it is not well known to what extent these recycling processes are implemented in practice. Therefore, the beneficial effects of recycling could be over- or as well underestimated based on the modeling assumptions. Nevertheless, the results show the importance of the implementation of effective recycling and waste treatment schemes.

The electric vehicles assessed in this study have been modeled with a rather small range of 120 km per battery charge. It is however probable that range is one of the most critical issues in terms of customers' acceptance of electric vehicles. Therefore, it is assumed that manufacturers will focus on the development of vehicles providing a higher range. Since the battery technology is the limiting factor for the provided range of today's EV, it is likely that research on the improvement of the batteries' storage capacity will come up with new technologies. Today, there are already a variety of different battery chemistries available for the application in electric vehicles. Therefore, it would be interesting to assess and compare life cycle impacts of different battery technologies, since the battery has a high contribution to the total impacts.

If the vehicles had been modeled with a larger driving range however, impacts of the battery production would have been higher. Additionally, the use phase energy consumption would increase as well due to the higher vehicle weight. Another consequence of a higher vehicle weight due to a heavier storage battery might be the need for a more powerful electric motor. This would in turn increase the impacts of the electric drivetrain. One aspect concerning the electric motor that was assumed to be critical is the permanent magnet. But there was no significant impact of the permanent magnet found in the assessment. The model of the permanent magnet is however very crude. Hence, it cannot be concluded whether the result is due to the poor model and impacts would be higher if assessed with better data, or whether the impacts had been overestimated in the first place.

The increase in battery and drivetrain weight could be outbalanced by means of lightweight construction of the glider in order to keep the use phase energy consumption on a constant level. In the assessment of different vehicle scenarios in this study, the implementation of lightweight construction has an overall positive effect on life cycle environmental impacts. However, the lightweight scenarios were calculated under the assumption of the same range of 120 km as in the baseline scenario (resulting in a lower battery weight) and with the same drivetrain components. In the assessment of the glider materialization per kilogram, the lightweight materials result in higher environmental impacts than the baseline. This leads to the question what the overall effect would be of downsizing the glider weight, while increasing the battery weight in favor of a higher range (whereas the use phase energy consumption would remain constant). It is assumed that the result would depend on the impact category, since the difference in impacts per kilogram of glider and battery material varies depending on the assessment method. For some impacts, such as global warming and fossil resource depletion, both lightweight glider options have higher impacts per kilogram than the battery. For other categories in contrast, for example human health impacts and damages to ecosystems, the battery results in higher impacts.

9.1 Data quality and further research needs

It has to be taken into account that the results presented in this study depend to a certain degree on the quality of the data used to establish the life cycle inventory of the product system. Uncertainties concerning the data quality arise on two levels. First, the accuracy of the data sources used to establish the inventories of the vehicle components in the foreground system has to be questioned. Second, the choice of Ecoinvent datasets to model production, use phase and disposal processes is another source for uncertainties.

The focus of the study has been on the materialization of the electric drivetrain and the vehicle glider. Concerning the electric drivetrain, the data quality of the material inventory seems to be satisfying, since it is based on supplier's data. Uncertainties arise however with respect to the choice of Ecoinvent datasets, because for some drivetrain materials no adequate datasets exist in the database. Therefore, several approximations were necessary. The greatest insecurity is seen in the model of the permanent magnet for the electric motor. Since there was no accurate data on permanent magnets available, it had to be modeled based on rather crude approximations. Hence, the material use and production processes, but also recycling procedures concerning permanent magnets would be an important topic for further research in the context of e-mobility. Additionally, it has to be mentioned that only one type of electric motor has been assessed in this study. Since different technologies of electric motors are applied in EV (with or without permanent magnets), it would be interesting to be able to compare the difference in environmental impacts between them.

Concerning the material inventory of the glider, the data quality for the baseline model seems to be sufficient, since several studies have reported similar values. It would be important though to compare and validate the model values with more accurate data from suppliers. For both lightweight glider models however, the material inventories are based on one single reference, because there was no more information available. Therefore, there is no possibility to verify the modeling assumptions. Generally, the amount of information on vehicle materialization and production processes found in scientific literature is rather poor. It should be a priority for future mobility research to acquire more precise data from suppliers on different vehicle materialization options. Further, the data on energy efforts for production processes is mainly based on rough assumptions. Therefore, it is another issue for further research to acquire more information on the whole vehicle production process.

Further, uncertainties arise concerning recycling and waste treatment processes of all vehicle components. Therefore, e-mobility research should investigate specifically on the end-of-life processes, in order to acquire better knowledge on existing recycling and waste treatment schemes and to come up with better data to model the environmental impacts of the end-of-life phase.

The assessment of impacts from the battery has not been in the main focus of this study. But, since the battery contributes a large share to the environmental impacts of electric vehicles, it would be interesting to compare different battery technologies against each other. There are a variety of different battery chemistries available that are applied in electric vehicles. Hence, the comparison of impacts from the production and disposal thereof could be another issue for further research.

Conclusion

Summarizing on the findings presented, electric vehicles have proven to be beneficial in terms of mitigating climate change impacts of personal mobility. In contrast, electric passenger cars cause higher impacts on human health and ecosystems compared to conventional cars. The vehicle infrastructure, i.e. the glider, the electric drivetrain and the storage battery, has a high contribution to these impacts. However, for some components uncertainties concerning the quality of data have to be considered. Nevertheless, if the vehicle components are subject to recycling processes at the EOL, human health impacts and damages on ecosystems can be diminished significantly.

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APPENDIX

I. Baseline Scenario: Comparison of Studies on vehicles' material content

Material contents of ICE vehicles in percent of total vehicle weight

	Reference Nr.	1	2	3	4a	4b	5	6	7	Average
Material Category	**Materials**									
ferrous metals	iron			8.00%				7.40%	10.70%	8.70%
ferrous metals	steel			49.00%	59.70%	65.40%		55.50%	54.50%	56.82%
	total ferrous metals	60.84%	66.70%	57.00%	59.70%	65.40%	60.20%	62.90%	65.20%	62.24%
non-ferrous metals	cast aluminum		4.90%						6.00%	5.45%
non-ferrous metals	wrought aluminum		1.40%						1.40%	1.40%
	total aluminum	5.47%	6.30%	10.00%	5.50%	4.90%	8.80%	7.70%	7.50%	7.02%
non-ferrous metals	magnesium		0.20%		0.04%	0.03%	1.80%	0.30%	0.20%	0.43%
non-ferrous metals	copper (&brass)	0.72%		1.10%	0.70%	0.60%	0.80%	1.60%	1.40%	0.99%
non-ferrous metals	lead				0.70%	0.60%	0.90%	1.10%		0.83%
non-ferrous metals	zinc			0.50%			0.60%	0.20%	0.40%	0.43%
non-ferrous metals	other metals							1.20%	1.10%	1.15%
	total without aluminum	2.52%	0.20%	1.60%	1.44%	1.23%	4.10%	4.40%	3.10%	2.32%
	total non-ferrous metals	7.99%	6.50%	11.60%	6.94%	6.13%	12.90%	12.10%	10.60%	9.34%
plastics	polyproplyene PP			9.50%	9.20%	7.80%				8.83%
plastics	polyethylene PE			2.00%	3.00%	2.50%				2.50%
plastics	polyethylene terephthalate				0.30%	0.30%				0.30%
plastics	polyamide PA			0.50%	0.50%	0.40%				0.47%
plastics	polyurethane PU			3.50%	2.40%	2.00%				2.63%
plastics	acrylonitrile butadine sterine ABS			1.00%	0.70%	0.60%				0.77%
plastics	polyvenil chloride PVC	0.15%		0.00%						0.08%
plastics	others			1.00%	2.20%	1.80%				1.67%
	total plastics	15.95%	5.90%	17.50%	18.30%	15.40%	8.80%	8.40%	7.50%	12.22%
various	glass (& ceramic)	2.80%	3.00%		3.20%	2.70%	3.20%	2.60%	3.00%	2.93%
various	glass fiber		0.60%							0.60%
various	resins		0.90%							0.90%
various	textiles				1.80%	1.60%		1.20%		1.53%
various	tyres & rubber	4.20%	4.30%		2.80%	2.30%	4.50%	4.50%	4.40%	3.86%
various	fuel/oil/lubricants	6.00%			4.00%	3.50%	3.50%	5.30%	6.00%	4.72%
various	insulation	1.50%								1.50%
various	paints	0.40%			2.90%	2.50%	0.50%	0.70%		1.40%
various	others	0.01%	12.00%				5.80%	2.20%	3.40%	4.68%
	total various	14.91%	20.80%	13.90%	15.06%	12.60%	17.50%	16.50%	16.80%	16.01%
electronic components	lights (LED)	0.01%								0.01%
electronic components	printed wiring board	0.19%								0.19%
	total electric components	0.20%								0.20%

References on vehicles' material content

Reference Nr	Reference	Titel
1	(Schweimer and Levin 2000)	Life Cycle Assement for Golf A4
2	(Tonn, Schexnayder et al. 2003)	An assessment of waste issues associated with the production of new, lightweight, fuel-efficient vehicles
3	(Giannouli, de Haan et al. 2007)	Waste from road transport: development of a model to predict waste from end-of-life and operation phases of road vehicles in Europe
4a,b[12]	(Leduc, Mongelli et al. 2010)	How can our cars become less polluting? An assessment of the environmental improvement potential of cars
5	(Weiss, Heywood et al. 2000)	On the road in 2020 - A life-cycle analysis of new automobile technologies
6	(U.S. Department of Energy 2010)	Transportation Energy Data Book
7	(Cheah 2010)	Cars on a Diet: The Material and Energy Impacts of Passenger Vehicle Weight Reduction in the U.S.

[12] 4a: material composition of petrol car, 4b: material composition of diesel car

II. Baseline Scenario: Inventory Glider production, Cut-off model

Process type	Materials	Ecoinvent Datasets	Unit	Amount/kg glider	WF	Input/kg glider
materials	total steel	Steel_mix*	kg	6.887E-01	1.00	6.887E-01
materials	total aluminum	Alu_mix*	kg	5.418E-03	1.00	5.418E-03
materials	magnesium	Magnesium, at plant/RER	kg	8.425E-04	1.01	8.509E-04
materials	copper	Copper, at regional storage/RER	kg	4.212E-03	1.00	4.212E-03
materials	lead	Lead, at regional storage/RER	kg	2.949E-03	1.01	2.978E-03
materials	zinc	Zinc, primary, at regional storage/RER	kg	1.685E-03	1.01	1.702E-03
materials	other metals	Copper, at regional storage/RER	kg	4.212E-03	1.01	4.255E-03
materials	total plastics	Plastic_mix*	kg	1.229E-01	1.00	1.229E-01
materials	glass	Flat glass, uncoated, at plant/RER	kg	4.002E-02	1.00	4.002E-02
materials	glass fiber	Glass fibre reinforced plastic, polyester resin, hand lay-up, at plant/RER U	kg	6.670E-03	1.10	7.337E-03
materials	resins	Epoxy resin, liquid, at plant/RER	kg	1.334E-02	1.10	1.467E-02
materials	textiles	Viscose fibres, at plant/GLO	kg	2.001E-02	1.10	2.201E-02
materials	tires & rubber	Synthetic rubber, at plant	kg	5.594E-02	1.01	5.650E-02
materials	oil & lubricants	Lubricating oil, at plant/RER	kg	3.752E-03	1.01	3.790E-03
materials	insulation	Polyurethane, flexible foam, at plant/RER	kg	1.334E-02	1.10	1.467E-02
materials	paints	Coating powder, at plant/RER U	kg	1.334E-02	1.10	1.467E-02
materials	lights (LED)	Light emitting diode, LED, at plant/GLO	kg	1.334E-04	1.00	1.334E-04
materials	printed wiring board	Printed wiring board, mixed mounted, unspec., solder mix, at plant/GLO	kg	2.535E-03	1.00	2.535E-03
processing	aluminum	Sheet rolling, aluminium/RER U	kg	5.418E-03	1.00	5.418E-03
processing	steel	Sheet rolling, steel/RER U	kg	6.887E-01	1.00	6.887E-01
processing	glass	Tempering, flat glass/RER	kg	4.002E-02	1.00	4.002E-02
processing	copper	Wire drawing, copper/RER U	kg	4.212E-03	1.00	4.212E-03
auxiliaries		Electricity, medium voltage, production UCTE, at grid/UCTE U	kWh			2.020E+00
auxiliaries		Heat, natural gas, at industrial furnace >100kW/RER U	MJ			2.095E+00
auxiliaries		Light fuel oil, burned in industrial furnace 1MW, non-modulating/RER U	MJ			5.946E-02
auxiliaries		Road vehicle plant/RER/I U	p			2.747E-10
auxiliaries		Tap water, at user/RER U	kg			3.039E+00
auxiliaries		Transport, freight, rail/RER U	tkm			5.002E-01
auxiliaries		Transport, lorry >16t, fleet average/RER U	tkm			5.002E-02
emissions to air		Heat, waste	MJ			7.268E+00
emissions to air		NMVOC, non-methane volatile organic compounds, unspecified origin	kg			4.531E-03
emissions to water		BOD5, Biological Oxygen Demand	kg			2.454E-05
emissions to water		COD, Chemical Oxygen Demand	kg			1.822E-04
emissions to water		Phosphate	kg			9.439E-07

* Definition of material mixes

Steel_mix	% of steel	Ecoinvent Datasets	Amount/kg material	WF	Input/kg material
conventional steel	78.0%	Reinforcing steel, at plant/RER	0.780	1.50	1.170
stainless steel	3.0%	Chromium steel 18/8, at plant/RER	0.030	1.25	0.038
high-strength steel	19.0%	Steel, low-alloyed, at plant/RER	0.190	1.25	0.238
Alu_mix	**% of aluminum**				
cast aluminum	80.0%	Aluminium, production mix, cast alloy, at plant/RER	0.800	1.10	0.880
wrought aluminum	20.0%	Aluminium, production mix, wrought alloy, at plant/RER	0.200	1.25	0.250
Plastic_mix	**% of plastic**				
polyproplyene PP	51.0%	Polypropylene, granulate, at plant/RER	0.510	1.01	0.515
polyethylene PE	15.0%	Polyethylene, LDPE, granulate, at plant/RER	0.150	1.01	0.152
polyethylene terephthalate	1.5%	Polyethylene terephthalate, granulate, amorphous, at plant/RER	0.015	1.01	0.015
polyamide PA	3.0%	Nylon 6, at plant/RER	0.030	1.01	0.030
polyurethane PU	15.0%	Polyurethane, flexible foam, at plant/RER	0.150	1.10	0.165
acrylonitrile butadine sterine ABS	4.5%	Acrylonitrile-butadiene-styrene copolymer, ABS, at plant/RER	0.045	1.01	0.045
others (PVC)	10.0%	Polyvinylchloride, at regional storage/RER	0.100	1.01	0.101

III. Baseline Scenario: Inventory Glider, System expansion and Substitution

a) Production of baseline glider, substitution model

Process type	Materials	Ecoinvent Datasets	Unit	Amount/kg glider	WF	Input/kg glider
materials	total steel	Steel_mix_primary*	kg	6.887E-01	1.00	6.887E-01
materials	total aluminum	Aluminium, primary, at plant/RER	kg	5.418E-03	1.10	5.960E-03
materials	magnesium	Magnesium, at plant/RER	kg	8.425E-04	1.01	8.509E-04
materials	copper	Copper, primary, mix/RER	kg	4.212E-03	1.00	4.212E-03
materials	lead	Lead, primary, at plant/RER	kg	2.949E-03	1.01	2.978E-03
materials	zinc	Zinc, primary, at regional storage/RER	kg	1.685E-03	1.01	1.702E-03
materials	other metals	Copper, primary, mix/RER	kg	4.212E-03	1.01	4.255E-03
materials	total plastics	Plastic_mix*	kg	1.229E-01	1.00	1.229E-01
materials	glass	Flat glass, uncoated, at plant/RER	kg	4.002E-02	1.00	4.002E-02
materials	glass fiber	Glass fibre reinforced plastic, polyester resin, hand lay-up, at plant/RER U	kg	6.670E-03	1.10	7.337E-03
materials	resins	Epoxy resin, liquid, at plant/RER	kg	1.334E-02	1.10	1.467E-02
materials	textiles	Viscose fibres, at plant/GLO	kg	2.001E-02	1.10	2.201E-02
materials	tires & rubber	Synthetic rubber, at plant	kg	5.594E-02	1.01	5.650E-02
materials	oil & lubricants	Lubricating oil, at plant/RER	kg	3.752E-03	1.01	3.790E-03
materials	insulation	Polyurethane, flexible foam, at plant/RER	kg	1.334E-02	1.10	1.467E-02
materials	paints	Coating powder, at plant/RER U	kg	1.334E-02	1.10	1.467E-02
materials	lights (LED)	Light emitting diode, LED_substitution	kg	1.334E-04	1.00	1.334E-04
materials	printed wiring board	Printed wiring board, mixed mounted, unspec., solder mix_substitution	kg	2.535E-03	1.00	2.535E-03
processing	aluminum	Sheet rolling, aluminium/RER U	kg	5.418E-03	1.00	5.418E-03
processing	steel	Sheet rolling, steel/RER U	kg	6.887E-01	1.00	6.887E-01
processing	glass	Tempering, flat glass/RER	kg	4.002E-02	1.00	4.002E-02
processing	copper	Wire drawing, copper/RER U	kg	4.212E-03	1.00	4.212E-03
auxiliaries		Electricity, medium voltage, production UCTE, at grid/UCTE U	kWh			2.020E+00
auxiliaries		Heat, natural gas, at industrial furnace >100kW/RER U	MJ			2.095E+00
auxiliaries		Light fuel oil, burned in industrial furnace 1MW, non-modulating/RER U	MJ			5.946E-02
auxiliaries		Road vehicle plant/RER/I U	p			2.747E-10
auxiliaries		Tap water, at user/RER U	kg			3.039E+00
auxiliaries		Transport, freight, rail/RER U	tkm			5.002E-01
auxiliaries		Transport, lorry >16t, fleet average/RER U	tkm			5.002E-02
emissions to air		Heat, waste	MJ			7.268E+00
emissions to air		NMVOC, non-methane volatile organic compounds, unspecified origin	kg			4.531E-03
emissions to water		BOD5, Biological Oxygen Demand	kg			2.454E-05
emissions to water		COD, Chemical Oxygen Demand	kg			1.822E-04
emissions to water		Phosphate	kg			9.439E-07

* Definition of material mixes

Steel_mix_primary	% of steel	Ecoinvent Datasets	Amount/kg material	WF	Input/kg material
conventional steel	78.0%	Steel, converter, unalloyed, at plant/RER	0.780	1.50	1.170
stainless steel	3.0%	Steel, converter, chromium steel 18/8, at plant/RER	0.030	1.25	0.038
high-strength steel	19.0%	Steel, converter, low-alloyed, at plant/RER	0.190	1.25	0.238
Plastic_mix	**% of plastic**				
polyproplyene PP	51.0%	Polypropylene, granulate, at plant/RER	0.510	1.01	0.515
polyethylene PE	15.0%	Polyethylene, LDPE, granulate, at plant/RER	0.150	1.01	0.152
polyethylene terephthalate	1.5%	Polyethylene terephthalate, granulate, amorphous, at plant/RER	0.015	1.01	0.015
polyamide PA	3.0%	Nylon 6, at plant/RER	0.030	1.01	0.030
polyurethane PU	15.0%	Polyurethane, flexible foam, at plant/RER	0.150	1.10	0.165
acrylonitrile butadine sterine ABS	4.5%	Acrylonitrile-butadiene-styrene copolymer, ABS, at plant/RER	0.045	1.01	0.045
others (PVC)	10.0%	Polyvinylchloride, at regional storage/RER	0.100	1.01	0.101

b) Disposal of baseline glider, substitution model

Avoided products	Ecoinvent Datasets	Unit	Amount/kg glider
Primary resources	steel_mix_primary	kg	6.887E-1*rec_f_Fe
Primary resources	Aluminium, primary, at plant/RER	kg	5.960E-3*rec_f_Al
Primary resources	Magnesium, at plant/RER U	kg	8.509E-4*rec_f_Al
Primary resources	Copper, primary, mix/RER U	kg	(4.212E-3+4.255E-3)*rec_f_Cu
Primary resources	Lead, primary, at plant/GLO U	kg	2.978E-3*rec_f_Pb
Primary resources	Zinc, primary, at regional storage/RER U	kg	1.702E-3*rec_f_Fe
Primary resources	Electricity, medium voltage, production UCTE, at grid/UCTE U	MJ	3.48*(1.229E-1+2.201E-2+1.467E-2)+3.02*5.650E-2+3.47*1.467E-2
Primary resources	Heat, natural gas, at boiler fan burner low-NOx non-modulating <100kW/RER U	MJ	7.03*(1.229E-1+2.201E-2+1.467E-2)+6.11*5.650E-2+7*1.467E-2

EOL processes	Ecoinvent Datasets	Unit	Amount/kg glider	Specifications
Recycling process	steel_mix_recycling	kg	6.887E-1*rec_f_Fe	
Recycling process	Steel, electric, un- and low-alloyed, at plant/RER U	kg	1.702E-3*rec_f_Fe	Recycling of Zinc
Recycling process	Aluminium, secondary, from old scrap, at plant/RER U	kg	5.418E-3*rec_f_Al	
Recycling process	Aluminium, secondary, from old scrap, at plant/RER U	kg	8.509E-4*rec_f_Al	Recycling of Magnesium
Recycling process	Copper, secondary, at refinery/RER U	kg	4.212E-3*rec_f_Cu	
Recycling process	Lead, secondary, at plant/RER U	kg	2.978E-3*rec_f_Pb	
Recycling process	Copper, secondary, at refinery/RER U	kg	4.255E-3*rec_f_Cu	Recycling of 'other metals'
Waste treatment	Disposal_PWB, mixed mounted, unspec., solder mix_substitution	kg	0.002535	
Waste treatment	Disposal_Light emitting diode, LED_substitution	kg	0.0001334	
Waste treatment	Disposal, plastics, mixture, 15.3% water, to municipal incineration/CH U	kg	1.229E-1+2.201E-2+1.467E-2	Plastic mix, viscose, epoxy resin Output: electricity 3.48 MJ/kg, heat 7.03 MJ/kg
Waste treatment	Disposal, glass, 0% water, to municipal incineration/CH U	kg	4.002E-2+(.337E-3/0.632)	Glass, glass fiber
Waste treatment	Disposal, rubber, unspecified, 0% water, to municipal incineration/CH U	kg	0.0565	Rubber Output: electricity 3.02 MJ/kg, heat 6.11 MJ/kg
Waste treatment	Disposal, used mineral oil, 10% water, to hazardous waste incineration/CH U	kg	0.00379	Lubricating oil
Waste treatment	Disposal, polyurethane, 0.2% water, to municipal incineration/CH U	kg	0.01467	Polyurethane Output: electricity 3.47MJ/kg, heat 7MJ/kg

*** Definition of steel recycling mix**

Steel_mix_recycling	% of steel	Ecoinvent Datasets	Amount/kg material	WF	Input/kg material
conventional steel	78.0%	Steel, electric, un- and low-alloyed, at plant/RER	0.780	1.50	1.170
stainless steel	3.0%	Steel, electric, chromium steel 18/8, at plant/RER	0.030	1.25	0.038
high-strength steel	19.0%	Steel, electric, un- and low-alloyed, at plant/RER	0.190	1.25	0.238

The inventory of the baseline glider in the substitution model is shown as an example for the difference in modeling between cut-off and substitution model. All other inventories have been modeled according to the same logic in the substitution model.

IV. Lightweight Scenario: Inventory Glider production, Cut-off model

		SCENARIO		LW PLASTIC	LW ALUMINUM		LW PLASTIC	LW ALUMINUM
Process type	**Materials**	**Ecoinvent Datasets**	**Unit**	**Amount/kg glider**	**Amount/kg glider**	**WF**	**Input/kg glider**	**Input/kg glider**
materials	steel_mix	Steel_mix*	kg	1.644E-01	1.255E-01	1.00	1.644E-01	1.255E-01
materials	cast aluminum	Aluminium, production mix, cast alloy, at plant/RER	kg	5.389E-02	3.931E-01	1.10	5.928E-02	4.324E-01
materials	wrought aluminum	Aluminium, production mix, wrought alloy, at plant/RER	kg	1.482E-01	1.457E-01	1.25	1.852E-01	1.822E-01
materials	magnesium	Magnesium, at plant/RER	kg	7.233E-02	3.717E-03	1.01	7.305E-02	3.754E-03
materials	copper	Copper, at regional storage/RER	kg	1.020E-02	1.068E-02	1.00	1.020E-02	1.068E-02
materials	lead	Lead, at regional storage/RER	kg	0.000E+00	0.000E+00	1.01	0.000E+00	0.000E+00
materials	zinc	Zinc, primary, at regional storage/RER	kg	5.127E-03	5.210E-03	1.01	5.178E-03	5.262E-03
materials	other metals	Copper, at regional storage/RER	kg	1.282E-02	1.302E-02	1.01	1.295E-02	1.315E-02
materials	titanium	Aluminium, primary, at plant/RER U	kg	2.371E-02	1.752E-02	1.10	2.609E-02	1.927E-02
materials	plastic_mix	plastic_mix	kg	1.253E-02	8.290E-02	1.00	1.253E-02	8.290E-02
materials	lexan	Polycarbonate, at plant/RER U	kg	1.186E-02	0.000E+00	1.10	1.304E-02	0.000E+00
materials	glass	Flat glass, uncoated, at plant/RER	kg	4.150E-02	3.027E-02	1.00	4.150E-02	3.027E-02
materials	glass fibre	Glass fibre reinforced plastic, polyester resin, hand lay-up, at plant/RER U	kg	3.557E-02	1.858E-02	1.10	3.913E-02	2.044E-02
materials	carbon fibre	carbon fibre	kg	1.423E-02	1.168E-02	1.10	1.565E-02	1.285E-02
materials	resins	Epoxy resin, liquid, at plant/RER	kg	2.537E-01	4.566E-02	1.10	2.791E-01	5.023E-02
materials	textiles	Viscose fibres, at plant/GLO	kg	1.923E-02	1.954E-02	1.10	2.115E-02	2.149E-02
materials	tyres&rubber	Synthetic rubber, at plant/RER U	kg	8.382E-02	3.737E-02	1.01	8.466E-02	3.775E-02
materials	oil&lubricants	Lubricating oil, at plant/RER	kg	8.644E-03	1.083E-02	1.01	8.730E-03	1.094E-02
materials	insulation	Polyurethane, flexible foam, at plant/RER	kg	1.282E-02	1.302E-02	1.10	1.410E-02	1.433E-02
materials	paints	Coating powder, at plant/RER U	kg	1.282E-02	1.302E-02	1.10	1.410E-02	1.433E-02
materials	lights (LED)	Light emitting diode, LED, at plant/GLO	kg	1.282E-04	1.302E-04	1.00	1.282E-04	1.302E-04
materials	printed wiring board	Printed wiring board, mixed mounted, unspec., solder mix, at plant/GLO	kg	2.435E-03	2.475E-03	1.00	2.435E-03	2.475E-03
processing	glass	Tempering, flat glass/RER	kg	4.150E-02	3.027E-02	1.00	4.150E-02	3.027E-02
processing	copper	Wire drawing, copper/RER U	kg	1.020E-02	1.068E-02	1.00	1.020E-02	1.068E-02
processing	steel	Sheet rolling, steel/RER U	kg	1.644E-01	1.255E-01	1.00	1.644E-01	1.255E-01
processing	aluminum	Sheet rolling, aluminium/RER U	kg	2.445E-01	6.146E-01	1.00	2.445E-01	6.146E-01
auxiliaries		Heat, natural gas, at industrial furnace >100kW/RER U	MJ				2.095E+00	2.095E+00
auxiliaries		Electricity, medium voltage, production UCTE, at grid/UCTE U	kWh				2.020E+00	2.020E+00

		SCENARIO		LW PLASTIC	LW ALUMINUM		LW PLASTIC	LW ALUMINUM
Process type	**Materials**	**Ecoinvent Datasets**	**Unit**	**Amount/kg glider**	**Amount/kg glider**	**WF**	**Input/kg glider**	**Input/kg glider**
auxiliaries		Light fuel oil, burned in industrial furnace 1MW, non-modulating/RER U	MJ				5.946E-02	5.946E-02
auxiliaries		Tap water, at user/RER U	kg				3.039E+00	3.039E+00
auxiliaries		Transport, lorry >16t, fleet average/RER U	tkm				5.002E-02	5.002E-02
auxiliaries		Transport, freight, rail/RER U	tkm				5.002E-01	5.002E-01
auxiliaries		Road vehicle plant/RER/I U	p				2.747E-10	2.747E-10
emissions to water		COD, Chemical Oxygen Demand	kg				1.822E-04	1.822E-04
emissions to water		BOD5, Biological Oxygen Demand	kg				2.454E-05	2.454E-05
emissions to water		Phosphate	kg				9.439E-07	9.439E-07
emissions to air		NMVOC, non-methane volatile organic compounds, unspecified origin	kg				4.531E-03	4.531E-03
emissions to air		Heat, waste	MJ				7.268E+00	7.268E+00

V. Electric Drivetrain: Inventory of Drivetrain components, Cut-off model

a) Production of Charger

Process type	Ecoinvent Datasets	Unit	Amount	WF	Input
materials	Aluminium, production mix, at plant/RER U	kg	4.200E-01	1.10	4.620E-01
materials	Aluminium, production mix, wrought alloy, at plant/RER U	kg	2.683E+00	1.25	3.354E+00
materials	Brass, at plant/CH U	kg	9.885E-02	1.01	9.984E-02
materials	Capacitor, film, through-hole mounting, at plant/GLO U	kg	3.446E-01	1.00	3.446E-01
materials	Chromium steel 18/8, at plant/RER U	kg	1.900E-01	1.25	2.375E-01
materials	Copper, at regional storage/RER U	kg	7.270E-01	1.00	7.270E-01
materials	Ferrite, at plant/GLO U	kg	7.550E-01	1.01	7.626E-01
materials	Polyester resin, unsaturated, at plant/RER U	kg	1.530E-01	1.10	1.683E-01
materials	Polystyrene, high impact, HIPS, at plant/RER U	kg	8.845E-02	1.01	8.933E-02
materials	Printed wiring board, surface mounted, unspec., solder mix, at plant/GLO U	kg	1.640E-01	1.00	1.640E-01
materials	Printed wiring board, through-hole, at plant/GLO U	kg	2.630E-01	1.00	2.630E-01
materials	Silicone product, at plant/RER U	kg	4.000E-03	1.01	4.040E-03
materials	Steel, low-alloyed, at plant/RER U	kg	3.310E-01	1.25	4.138E-01
materials	Aluminium oxide, at plant/RER U	kg	6.000E-03	1.01	6.060E-03
processing	Selective coating, aluminium sheet, nickel pigmented aluminium oxide/SK U	m2	3.961E-04	1.00	3.961E-04
processing	Zinc coating, coils/RER U	m2	1.631E-02	1.00	1.631E-02
processing	Production efforts, resistors/GLO U	kg	6.000E-03	1.00	6.000E-03
processing	Sheet rolling, steel/RER U	kg	7.331E-01	1.00	7.331E-01
processing	Wire drawing, copper/RER U	kg	7.550E-01	1.00	7.550E-01
processing	Sheet rolling, aluminium/RER U	kg	3.454E+00	1.00	3.454E+00
processing	Electricity, low voltage, at grid/CH U	kWh			confidential
processing	Heat, wood pellets, at furnace 50kW/CH U	MJ			confidential

b) Production of DC/DC-Converter

Process type	Ecoinvent Datasets	Unit	Amount	WF	Input
materials	Aluminium oxide, at plant/RER U	kg	2.500E-02	1.01	2.525E-02
materials	Aluminium, production mix, at plant/RER U	kg	2.445E+00	1.10	2.690E+00
materials	Brass, at plant/CH U	kg	6.000E-02	1.01	6.060E-02
materials	Capacitor, film, through-hole mounting, at plant/GLO U	kg	1.360E-01	1.00	1.360E-01
materials	Chromium steel 18/8, at plant/RER U	kg	9.520E-02	1.25	1.190E-01
materials	Copper, at regional storage/RER U	kg	5.400E-01	1.00	5.400E-01
materials	Ferrite, at plant/GLO U	kg	5.097E-01	1.01	5.148E-01
materials	Polyester resin, unsaturated, at plant/RER U	kg	1.000E-01	1.10	1.100E-01
materials	Polystyrene, high impact, HIPS, at plant/RER U	kg	2.450E-02	1.01	2.475E-02
materials	Printed wiring board, surface mounted, unspec., solder mix, at plant/GLO U	kg	2.669E-01	1.00	2.669E-01
materials	Printed wiring board, through-hole, at plant/GLO U	kg	1.003E-01	1.00	1.003E-01
materials	Silicone product, at plant/RER U	kg	2.490E-03	1.01	2.515E-03
materials	Steel, low-alloyed, at plant/RER U	kg	1.800E-01	1.25	2.250E-01
processing	Production efforts, resistors/GLO U	kg	2.500E-02	1.00	2.500E-02
processing	Sheet rolling, aluminium/RER U	kg	2.690E+00	1.00	2.690E+00
processing	Sheet rolling, steel/RER U	kg	3.440E-01	1.00	3.440E-01
processing	Wire drawing, copper/RER U	kg	5.400E-01	1.00	5.400E-01
processing	Electricity, low voltage, at grid/CH U	kWh			confidential
processing	Heat, wood pellets, at furnace 50kW/CH U	MJ			confidential

c) Production of Electric Motor

	Motor type			Brusa	Brusa	Perm	Perm
Process type	**Ecoinvent Datasets**	**Unit**	**WF**	**Amount**	**Input**	**Amount**	**Input**
material	Aluminium oxide, at plant/RER U	kg	1.01	1.710E-01	1.727E-01	1.62E-01	1.63E-01
material	Aluminium, production mix, wrought alloy, at plant/RER U	kg	1.25	1.027E+01	1.283E+01	8.37E+00	1.05E+01
material	Brass, at plant/CH U	kg	1.01	1.880E-01	1.899E-01	0.00E+00	0.00E+00
material	Chromium steel 18/8, at plant/RER U	kg	1.25	5.149E+00	6.436E+00	2.36E+00	2.95E+00
material	Copper, at regional storage/RER U	kg	1.00	7.150E+00	7.150E+00	5.58E+00	5.58E+00
material	NdFeB**	kg	1.10	1.800E+00	1.980E+00	9.30E-01	1.02E+00
material	Nylon 6, at plant/RER U	kg	1.01	2.500E-01	2.525E-01	2.36E-01	2.39E-01
material	Polyester resin, unsaturated, at plant/RER U	kg	1.10	1.000E-01	1.100E-01	9.45E-02	1.04E-01
material	Polystyrene, high impact, HIPS, at plant/RER U	kg	1.01	5.000E-02	5.050E-02	4.73E-02	4.77E-02
material	Printed wiring board, surface mounted, unspec., solder mix, at plant/GLO U	kg	1.00	4.000E-02	4.000E-02	3.78E-02	3.78E-02
material	Printed wiring board, through-hole, at plant/GLO U	kg	1.00	4.500E-02	4.500E-02	4.25E-02	4.25E-02
material	Steel, low-alloyed, at plant/RER U	kg	1.25	2.865E+01	3.581E+01	1.31E+01	1.64E+01
processing	Production efforts, resistors/GLO U	kg	1.00	1.710E-01	1.710E-01	1.62E-01	1.62E-01
processing	Selective coating, aluminium sheet, nickel pigmented aluminium oxide/SK U	m2	1.00	5.549E-03	5.549E-03	0	0.00E+00
processing	Sheet rolling, steel/RER U	kg			4.225E+01		1.94E+01
processing	Wire drawing, copper/RER U	kg	1.00	7.150E+00	7.150E+00	5.58E+00	5.58E+00
processing	Sheet rolling, aluminium/RER U	kg			1.283E+01		1.05E+01
processing	Electricity, low voltage, at grid/CH U	kWh			confidential		0
processing	Electricity, low voltage, production UCTE, at grid/UCTE U	kWh			0		confidential
processing	Heat, wood pellets, at furnace 50kW/CH U	MJ			confidential		0
processing	Heat, natural gas, at industrial furnace >100kW/RER U	MJ			0		confidential
processing	Electricity, medium voltage, at grid/CN U	kWh			2.652E+01*		1.53E+01*
processing	Heat, natural gas, at industrial furnace >100kW/RER U	MJ			2.235E+01*		1.29E+01*

* Approximation of energy efforts for motor production with data on ABB AC Low voltage cast iron motor (M3BP 315) (ABB 2002)
Electricity = 0.492 kWh/kg
Heat = 0.415 kWh/kg

** see Appendix IV d)

d) Assumptions for production of NdFeB permanent magnet

$1\ Nd_2O_3 + 7\ Fe + 0.5\ B_2O_3 \rightarrow 1\ Nd_2Fe_{14}B$

Molecular Formula	Mol	g	w%	kg
$Nd_2Fe_{14}B$	1	1081.19		1
Nd_2O_3	1	336.48	31.12%	0.311
B_2O_3	0.5	34.81	3.22%	0.032
Fe	14	781.9	72.32%	0.723

Process type	Ecoinvent Datasets	Unit	Input/kg	Remarks
Nd material	Neodymium oxide, at plant/CN U	kg	0.311	
Fe material	Pig iron, at plant/GLO U	kg	0.723	
B material	Boric oxide, at plant/GLO U	kg	0.032	
processing	Aluminium, primary, liquid, at plant/RER U	kg	0.343	same processing of Nd and B assumed as for primary Al production
processing	Aluminium oxide, at plant/RER U	kg	-0.660	subtraction of Al materials from processing

e) Production of AC/DC-Inverter

Process type	Ecoinvent Datasets	Unit	Amount	WF	Input
materials	Aluminium oxide, at plant/RER U	kg	2.800E-02	1.01	2.828E-02
materials	Aluminium, production mix, at plant/RER U	kg	4.600E-02	1.10	5.060E-02
materials	Aluminium, production mix, wrought alloy, at plant/RER U	kg	5.004E+00	1.25	6.255E+00
materials	Brass, at plant/CH U	kg	3.230E-01	1.01	3.262E-01
materials	Capacitor, film, through-hole mounting, at plant/GLO U	kg	8.920E-01	1.00	8.920E-01
materials	Copper, at regional storage/RER U	kg	1.450E+00	1.00	1.450E+00
materials	Ferrite, at plant/GLO U	kg	3.010E-01	1.01	3.040E-01
materials	Polyester resin, unsaturated, at plant/RER U	kg	2.206E-01	1.10	2.427E-01
materials	Polyethylene, LDPE, granulate, at plant/RER	kg	2.400E-02	1.01	2.424E-02
materials	Polystyrene, high impact, HIPS, at plant/RER U	kg	1.980E-01	1.01	2.000E-01
materials	Printed wiring board, surface mounted, unspec., solder mix, at plant/GLO U	kg	6.810E-01	1.00	6.810E-01
materials	Printed wiring board, through-hole, at plant/GLO U	kg	2.283E-02	1.00	2.283E-02
materials	Silicone product, at plant/RER U	kg	4.000E-03	1.01	4.040E-03
materials	Steel, low-alloyed, at plant/RER U	kg	3.240E-01	1.25	4.050E-01
processing	Production efforts, resistors/GLO U	kg	2.800E-02	1.00	2.800E-02
processing	Selective coating, aluminium sheet, nickel pigmented aluminium oxide/SK U	m2	2.815E-03	1.00	2.815E-03
processing	Sheet rolling, aluminium/RER U	kg	6.306E+00	1.00	6.306E+00
processing	Sheet rolling, steel/RER U	kg	4.050E-01	1.00	4.050E-01
processing	Wire drawing, copper/RER U	kg	1.450E+00	1.00	1.450E+00
processing	Electricity, low voltage, at grid/CH U	kWh			confidential
processing	Heat, wood pellets, at furnace 50kW/CH U	MJ			confidential

f) Production of Power Distribution Unit (PDU)

Process type	Dataset	Unit	Amount	WF	Input
materials	Aluminium oxide, at plant/RER U	kg	1.150E-01	1.01	1.162E-01
materials	Aluminium, production mix, wrought alloy, at plant/RER U	kg	1.800E+00	1.25	2.250E+00
materials	Brass, at plant/CH U	kg	2.990E-01	1.01	3.020E-01
materials	Copper, at regional storage/RER U	kg	8.850E-01	1.00	8.850E-01
materials	Polystyrene, high impact, HIPS, at plant/RER U	kg	5.900E-02	1.01	5.959E-02
materials	Printed wiring board, surface mounted, unspec., solder mix, at plant/GLO U	kg	4.680E-01	1.00	4.680E-01
materials	Resistor, SMD type, surface mounting, at plant/GLO U	kg	3.000E-02	1.00	3.000E-02
materials	Steel, low-alloyed, at plant/RER U	kg	2.480E-01	1.25	3.100E-01
processing	Production efforts, resistors/GLO U	kg	1.150E-01	1.00	1.150E-01
processing	Sheet rolling, aluminium/RER U	kg	2.250E+00	1.00	2.250E+00
processing	Sheet rolling, steel/RER U	kg	3.100E-01	1.00	3.100E-01
processing	Wire drawing, copper/RER U	kg	8.850E-01	1.00	8.850E-01
processing	Electricity, low voltage, at grid/CH U	kWh			confidential
processing	Heat, wood pellets, at furnace 50kW/CH U	MJ			confidential