Life Cycle Sustainability Assessment at an Automotive Company

Identification and Implementation of Product Sustainability Improvement Potential

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Abstract

Purpose: The assessment and improvement of sustainability impacts of products is integral to achieve sustainable development. A way to assess the impact of a product over its entire life cycle on all three dimensions of sustainability (environment, economy and society) is Life Cycle Sustainability Assessment (LCSA). LCSA suggests to carry out three complementary assessments: Life Cycle Assessment (LCA), Life Cycle Costing (LCC) and Social Life Cycle Assessment (S-LCA). While LCA is widely applied across various industries, the use of LCSA at industrial companies is still scarce. This is, inter alia, due to the challenges in data acquisition in S-LCA and missing support in decision making based on LCSA results as well as their implementation in the decision processes at companies. This thesis aims at remediating these shortcomings to achieve its overarching goal: Increasing the applicability of LCSA to identify and implement product sustainability improvement potential. For this, three research objectives were defined to address the main impediments. First, an approach to select social indicators and to focus primary data collection in S-LCA. Second, a method to interpret LCSA results for decision makers. Third, a mechanism to integrate LCSA based results into product-related decision making at an automotive company.

Methods: To put the LCSA framework into practice at an automotive company, it was applied to the component level of a vehicle to assess the impact of every component over its life cycle. In the proposed framework, criticality scores for every LCSA dimension were defined. They were calculated based on the relative performance of a component in the respective LCSA dimension (LCA, LCC and S-LCA) to the rest of the components of the assessed vehicle. This conceptual approach formed the bearing for the integration of the individual research objectives. The three research objectives were addressed in individual publications. The results of these separate papers were combined to increase the applicability of the conceptual approach of the LCSA framework. Multi-Regional Input-Output (MRIO) databases were assessed regarding their possible use to address the first research objective. The findings were applied to enable an assessment of the S-LCA dimension in the LCSA framework. A study on the weighting of sustainability dimensions by decision makers in an automotive company was carried out to tackle the second research objective. The results were used to determine an overall LCSA impact result for every component of a vehicle. To address the third research objective, the customer added value of product sustainability features was investigated. The results were used to create a mechanism that enabled the integration of the assessment results of the LCSA framework alongside other indicators in product-related decision processes at an automotive company.

Results: Supply chain analysis based on MRIO analysis did not offer substantial support for focusing primary data collection in S-LCA. Therefore, a material-based approach to assess the social risk related to material production was developed and used to determine component specific S-LCA criticality points. The introduction of weights enabled a combined assessment within the operationalized LCSA framework, i.e. the calculation of overall LCSA criticality points. From the determined customer added value, a Product Sustainability Budget (PSB) was deduced. The PSB could be used to identify the optimal set of improvement measures within defined economic boundaries that are acceptable to a company.

Conclusion: It was shown in this thesis that the applicability of the LCSA framework could be increased by separately addressing the identified research needs and bringing the results together. Through the combined findings progress was made to support practitioners at an automotive company to identify the components with the highest product sustainability impacts and therefore greatest improvement potential. They also were provided with a mechanism - the PSB - to implement the improvement potential in the product-related decision process. There still remained several challenges that limited the degree of applicability of the LCSA framework. It was found that the data availability for S-LCA still remains a major challenge. The current LCSA framework allocates the impacts of the use phase of the entire vehicle via mass allocation to the individual components. Additional investigation of the relationship between components and their impact on the use phase of a vehicle is recommended. Also, the PSB still has to prove itself under real conditions.

Zusammenfassung

Ziel: Die Bewertung und Verbesserung der Nachhaltigkeitsauswirkungen von Produkten ist ein integraler Bestandteil einer nachhaltigen Entwicklung. Eine Möglichkeit, die Auswirkungen eines Produkts auf alle drei Dimensionen der Nachhaltigkeit (Umwelt, Wirtschaft und Soziales) zu bewerten, bietet das Life Cycle Sustainability Assessment (LCSA). LCSA schlägt die komplementäre Durchführung von drei lebenszyklusbasierten Bewertungen vor: Life Cycle Assessment (LCA), Life Cycle Costing (LCC) und Social Life Cycle Assessment (S-LCA). Während LCA bereits in verschiedensten Industriesektoren Anwendung findet, wird LCSA kaum angewendet. Dies ist, unter anderem, den Herausforderungen in der Datenbeschaffung für S-LCA sowie der fehlenden Unterstützung in der Interpretation und Berücksichtigung von LCSA-Ergebnissen in produktbezogenen Entscheidungsprozessen in Unternehmen geschuldet. Die vorliegende Dissertation hat sich vorgenommen, diese Herausforderungen zu überwinden, um ihr übergeordnetes Ziel zu erreichen: die Erhöhung der Anwendbarkeit von LCSA, um Potentiale zur Verbesserung der Produktnachhaltigkeit identifizieren und umsetzen zu können. Es wurden daher drei Forschungsziele definiert, die die größten Herausforderungen adressierten. Das erste Ziel war die Entwicklung eines Ansatzes, um soziale Indikatoren zu identifizieren und die Primärdatenerhebung für S-LCA zu fokussieren. Das zweite Ziel war die Entwicklung einer Methode, um LCSA Ergebnisse für Entscheidungsträger zu interpretieren. Das dritte Ziel war die Schaffung eines Mechanismus, mit dem die LCSA Ergebnisse in den produktbezogenen Entscheidungsprozessen im Unternehmen implementiert werden können.

Methoden: Um LCSA in einem Automobilunternehmen in die Praxis umzusetzen, wurde es auf die Komponentenebene eines Fahrzeugs angewandt, damit die Auswirkungen jeder Komponente über ihren Lebenszyklus bewertet werden konnten. In dem vorgeschlagenen Modell zur Umsetzung wurden Kritikalitätspunkte für jede der drei LCSA Dimensionen (LCA, LCC und S-LCA) definiert. Diese wurden auf Basis der relativen Auswirkung einer Komponente in einer LCSA Dimension im Verhältnis zu den übrigen Komponenten im gesamten Fahrzeug berechnet. Dieser konzeptuelle Ansatz diente als Leitlinie für die Einordnung der drei separaten Forschungsziele unter das Forschungsziel der Gesamtarbeit. Die Bearbeitung der drei Forschungsziele erfolgte durch alleinstehende Veröffentlichungen. Deren Ergebnisse wurden kombiniert, um die Anwendbarkeit des konzeptuellen LCSA Ansatzes zu verbessern. Multi-Regionale Input-Output (MRIO) Datenbanken wurden hinsichtlich ihrer möglichen Eignung zur Erreichung des ersten Forschungsziels untersucht. Die Ergebnisse aus dieser Untersuchung wurden genutzt, um die S-LCA Bewertung innerhalb von LCSA zu befähigen. Eine Studie zur Gewichtung von Nachhaltigkeitsdimensionen durch Entscheidungsträger wurde in einem Automobilunternehmen durchgeführt, um das zweite Forschungsziel zu erreichen. Die Ergebnisse der Studie wurden genutzt, um für jede Komponente eine übergreifende LCSA Kritikalitätspunktzahl bestimmen zu können. Um das dritte Forschungsziel zu adressieren, wurde der Kundenmehrwert von Nachhaltigkeits-Features in einem Fahrzeug ermittelt. Die Ergebnisse aus der Bearbeitung dieses Forschungsziels wurden genutzt, um einen Mechanismus zu entwerfen, der die Integration von LCSA Ergebnissen neben etablierten Indikatoren im produktbezogenen Entscheidungsprozess in einem Automobilunternehmen ermöglichen sollte.

Ergebnisse: Lieferkettenanalyse auf Basis von MRIO Datenbanken bot keinen substanziellen Vorteil für die Fokussierung der Primärdatenerhebung für S-LCA. Daher wurde ein materialbezogener Ansatz zur Bewertung der sozialen Risiken, die im Zusammenhang mit der Materialbereitstellung stehen, entwickelt. Die Ergebnisse dieses Ansatzes wurden eingesetzt, um die komponentenspezifischen S-LCA Kritikalitätspunkte zu bestimmen. Die Einführung von Gewichtungen ermglichte die kombinierte Bewertung innerhalb der operationalisierten LCSA, das heißt die Berechnung der übergreifenden LCSA Kritikalitätspunkte pro Komponente. Aus dem Kundenmehrwert von Produktnachhaltigkeits-Features wurde ein Produktnachhaltigkeitsbudget (PNB) abgeleitet. Das PNB konnte genutzt werden, um das optimale Set an Verbesserungsmaßnahmen für die Produktnachhaltigkeit zu identifizieren, das innerhalb des akzeptablen ökonomischen Rahmens des Unternehmens lag.

Fazit: Es wurde im Rahmen dieser Dissertation gezeigt, dass die Anwendbarkeit von LCSA durch die Beantwortung von drei separaten Forschungsbedarfen und der Kombination der Ergebnisse erhöht werden konnte. Durch die Zusammenführung der Erkenntnisse konnten Fortschritte in der Unterstützung von Anwendern bei einem Automobilunternehmen erzielt werden, die Komponenten mit den höchsten Auswirkungen - und damit den höchsten Verbesserungspotenzialen - in Bezug auf die Produktnachhaltigkeit zu identifizieren. Zusätzlich wurde ihnen ein Mechanismus - das PNB - zur Verfügung gestellt, mit dem sie die Verbesserungspotenziale in den produktbezogenen Entscheidungsprozessen implementieren konnten. Es bleiben dennoch einige Herausforderungen offen, die die Anwendbarkeit von LCSA einschränken. Es wurde in Rahmen dieser Arbeit erneut bestätigt, dass die Datenverfügbarkeit für S-LCA eine der größten Herausforderungen bleibt. Der LCSA Ansatz, wie er in dieser Arbeit vorgestellt wurde, teilte die Gesamtfahrzeugauswirkungen in der Nutzungsphase per Gewichtsanteil auf die Komponenten auf. Zusätzliche Untersuchungen des Zusammenhangs einzelner Komponenten und den Auswirkungen in der Nutzungsphase des Gesamtfahrzeugs werden daher empfohlen. Des Weiteren muss sich das Konzept des PNB noch unter realen Bedingungen beweisen.

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List of Abbreviations

BEV	Battery-electric Vehicle
BMW	Bayerische Motorenwerke
CBCA	Choice-Based Conjoint Analysis
CEO	Chief Executive Officer
CIC	Cumulated Intermediate Consumption
CSR	Corporate Social Responsibility
DB	Database
e.g.	For example (Latin: *exempli gratia*)
EoL	End of Life
EV	Electric Vehicle
GWP	Global Warming Potential
ICE	Internal Combustion Engine
i.e.	That means (Latin: *id est*)
ILO	International Labour Organization
IPP	Integrated Product Policy
ISO	International Organization for Standardization
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCSA	Life Cycle Sustainability Assessment
Mio.	Million
MRIO	Multi-Regional Input-Output
OEM	Original Equipment Manufacturer

PSB	Product Sustainability Budget
PSI	Product Sustainability Index
RDE	Real Driving Emissions
RO	Research Objective
RS	Risk Score
SC	Supply Chain
SETAC	Society of Environmental Toxicology and Chemistry
SHDB	Social Hotspot Database
S-LCA	Social Life Cycle Assessment
SOLCA	Social Organizational Life Cycle Assessment
SUV	Sports Utility Vehicle
SVR	Sustainable Value Report
tbd.	To be determined
UAE	United Arab Emirates
UNEP	United Nations Environmental Programme
USA	Unites States of America
USGS	United States Geological Survey
WCED	World Commission on Economic Development
WTP	Willingness To Pay

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Publications

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- Tarne, P., Lehmann, A., Finkbeiner, M. (2018a). A comparison of Multi-Regional Input-Output databases regarding transaction structure and supply chain analysis. *Journal of Cleaner Production*, 196, 1486–1500. https://doi.org/10.1016/j.jclepro.2018.06.082
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- Tarne, P., Lehmann, A., Finkbeiner, M. (2018b). Introducing weights to life cycle sustainability assessment — how do decision-makers weight sustainability dimensions? *The International Journal of Life Cycle Assessment.* https://doi.org/10.1007/s11367-018-1468-2
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- Tarne, P., Lehmann, A., Kantner, M., Finkbeiner, M. (2018c). Introducing a product sustainability budget at an automotive company — one option to increase the use of LCSA results in decision-making processes. *The International Journal of Life Cycle Assessment.* https://doi.org/10.1007/s11367-018-1576-z
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1 Introduction & Background

This section gives an introduction to the concepts of sustainability and product sustainability (section 1.1) and the status of sustainability in the automotive industry (section 1.2). The background on Life Cycle Sustainability Assessment (LCSA) as a framework to assess product sustainability is given in section 1.3. Section 1.4 outlines the state of the art of LCSA and its application in the automotive industry to identify research gaps that need to be addressed in order to put LCSA into practice at an automotive company. The goal of this thesis and the corresponding research objectives are presented in section 1.5 while the structure is laid out in section 1.6.

1.1 The Concepts of Sustainability and Product Sustainability

In 1972, the "Limits to Growth" of humanity and its way of life were demonstrated as the exponential increase in population and economic growth is supported by a limited system, the planet earth (Meadows et al., 1972). In a follow up 30 years later, Meadows et al. reinforced their message as according to their model calculations, humanity was straining earth beyond its capacity limits since the 1980s (Meadows et al., 2007). The concept of sustainability is a concept that can enable humanity to not deprive itself of its livelihood. The most prominent definition of a sustainable development comes from the so-called Brundtland Report which characterized it as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987). This rather abstract concept of sustainability was adopted in 1992 at the World Summit of the United Nations Environmental Programme (UNEP) in Rio de Janeiro as central concept to achieve a healthy and productive life style in keeping with nature's boundaries (Report Of The United Nations Conference On And Development, 1992). This established the concept of sustainability at the political level, for the implementation in economic realm, a more concrete guidance was needed. This was partially achieved by the introduction of the 'Triple Bottom Line' that postulates that sustainability is achieved by the combined consideration of the three so-called pillars — or dimensions — of sustainability: economy, environment and society (Elkington, 1998). There are different approaches how to jointly evaluate the impacts on these three dimensions of sustainability. They can be differentiated into "strong" and "weak" interpretations of sustainability (JRC, 2012). The approach of Elkington would be characterized as a weak approach as positive impacts on two dimensions can compensate for negative effects in the third dimension. That means, for instance, ecosystem services can be substituted for increased economic gain and improved social conditions (Gibbs et al., 1998). JRC (2012) propose a hierarchical approach in order to respect that the opportunities to grow are limited by natural boundaries. They consider the environmental dimension to be of pivotal importance as all production processes take place within the ecosphere of the planet. The environmental dimension is followed by the social dimension and then by the economic dimension. That means, ecosystem services cannot be substituted by improvements in other dimensions (Gibbs et al., 1998). Figure 1 shows the two concepts vis-à-vis.

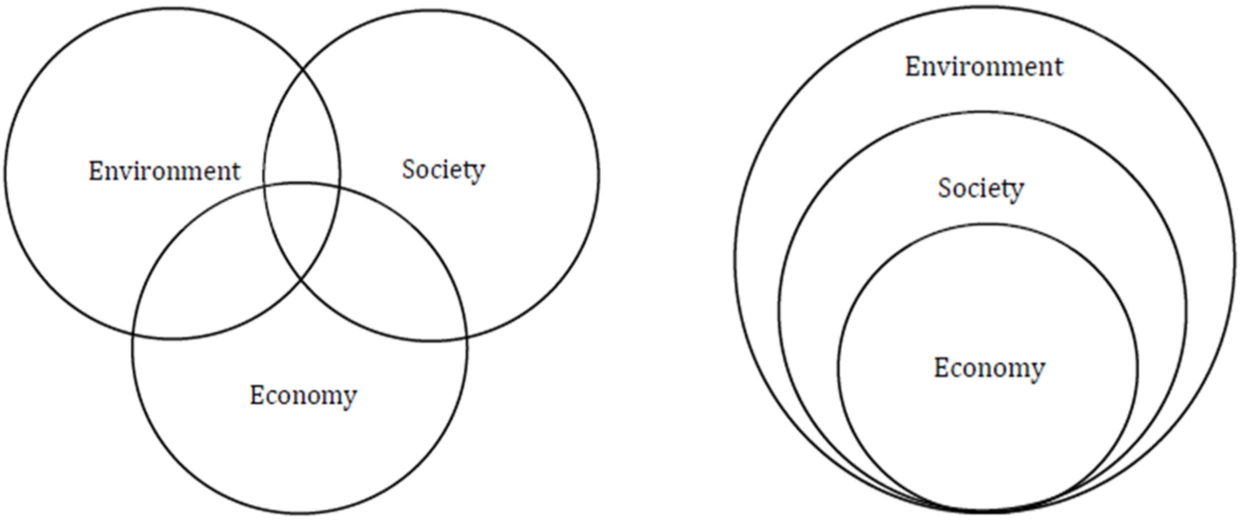

Figure 1: Weak and strong concept of sustainability. In the weak interpretation (left), the dimensions are partly independent from one another. In the strong interpretation, the economic activities are subordinate to the social dimension, which is limited by the environmental capacity (JRC, 2012).

Next to the widely adopted concept containing the three sustainability dimensions, there are other concepts including additional "institutional" dimension (e.g. Jörissen et al. (1999) and Spangenberg et al. (2002)) and concepts with up to five (Seghezzo, 2009) or even more dimensions (Čuček et al., 2012). Singh et al. (2009) give an overview of existing sustainability assessment methodologies.

The manufacturing of products[1] is always connected to environmental impacts like resource use, land use and the pollution of environmental media (water, soil, air) through emissions and waste. Furthermore, products have an impact on economy and society as their production creates jobs, generates revenue and they add value for consumers by satisfying their needs. Therefore, products are connected to all three dimensions of sustainability. The importance of improving product sustainability has been recognized, inter alia, by the European Commission that launched the Integrated Product Policy (IPP) to address product environmental impacts (Tukker et al., 2006). The Environmental Impact of Products (EIPRO) study identified cars, food, heating and house building to be the product groups with the highest environmental impacts across seven different, fully evaluated studies (Tukker & Jansen, 2006). Improving product sustainability within the automotive industry would therefore

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[1] "Product" in this thesis is defined according to the ISO 26000: "Any goods or service offered to members of the public either by sales or otherwise." (ISO 26000, 2010)

help to further support sustainable development. As the manufacturing and distribution of products is not performed by a single company at a single site but is embedded in an entire supply chain, approaches to the assessment and improvement of sustainability in general, and in product sustainability in particular, should take the entire supply chain into consideration (Srivastava, 2007; Miemczyk et al., 2012). Especially for companies that manage well-known brands and therefore operate in an exposed position, the integral management of the supply chain is paramount as environmental and social violations in the supply chain fall back on the central company (Koplin et al., 2007).

Despite the increasing importance of sustainability in the supply chain, there is still no clear understanding of how sustainability in the supply chain should best be captured and assessed (Freidberg, 2013).

Even though many publications deal with the concept and assessment of product sustainability, a clear and universal definition is still missing. In the course of this thesis, the term "product sustainability" will refer to the adapted definition of sustainable consumption and production of the UN (2011):

> *"Sustainable products bring a better quality of life while minimizing negative environmental, economic and social impacts over its life cycle so as not to jeopardize the needs of future generation."*

This definition implies a relative evaluation of the sustainability of a product, meaning that it is assessed whether a product or concept of a product is more sustainable or less sustainable than a respective alternative. An absolute evaluation of the sustainability of a product, i.e. whether a product contributes to a sustainable development at all, is not possible when applying this relative approach. That means, this research follows the path of relative sustainability assessment, while absolute sustainability evaluation, i.e. statements on whether a vehicle is sustainable or not, lies outside its scope.

1.2 Sustainability in the Automotive Sector

When looking at the status of sustainability in the automotive industry, the distinction between sustainability and product sustainability has to be made as well. Companies in the automotive industry can have measures and strategies in place that address the corporate level of sustainability which are usually subsumed under the term of Corporate Social Responsibility (CSR). They can also have measures and strategies in place that aim at improving the sustainability of their products. Sustainability at the corporate level is well understood within the automotive industry and broadly implemented (Sukitsch et al., 2015; Rodrigues Vaz et al., 2017). This is exemplified by the fact that most European Original Equipment Manufacturers (OEM) publish sustainability reports (Sukitsch et al., 2015). Regarding the information disclosed in sustainability reports and the indicators that are compiled to assess sustainability performance, companies usually follow the guidelines of the Global Reporting Initiative (Global Reporting Initiative, 2015, 2018). In addition to publishing information on their own sustainability measures, OEM generally also demand transparency and improvement of environmental performance of their suppliers. This is especially the case when the OEM itself has implemented an environmental management system like ISO 14001 (González et al., 2008). One of the advantages of the adoption of sustainability at the corporate level is the potential for "green marketing", i.e. positioning the company as ambitious actor in terms of sustainability and influencing its image accordingly (Simão & Lisboa, 2017). This positive image can positively influence the revenue of a company — for the automotive industry, 5.7% of revenues can be attributed to the sustainability image (Biesalski & Company, 2014). The existing approaches that try to assess the impacts on the three dimensions of sustainability in the automotive industry target the company level (Jasiński et al., 2016; Azevedo & Barros, 2017; Fritz et al., 2018). The approach by Azevedo & Barros (2017) developed a tool that took data for assessing the impacts on the triple bottom line for UK based automotive manufacturers. Jasiński et al. (2016) selected a set of sustainability assessment criteria from the literature and in combination with the results of 24 interviews with automotive experts they suggested 26 midpoint and 9 end-point environmental, resource, social and economic impact categories. The method of Fritz et al. (2018) focused more on the assessment of supply chain sustainability in the automotive industry. They propose to assess governance, environmental and social indicators on company level. Regarding product sustainability there is no clear consensus on what the automotive sustainability assessment criteria are (Jasiński et al., 2016). Mostly, only environmental aspects are considered except for the Product Sustainability Index (PSI) that has been started by Ford in 2006 (Jasiński et al., 2016). The PSI contains five environmental, two social and one economic indicator but has not been developed further (Schmidt & Taylor, 2006). Common environmental practices in the automotive industry to improve a product's impacts are (1) greenhouse gas minimization, (2) Life Cycle Assessment (LCA), (3) Eco-innovations, (4) Material selection, (5) Cleaner production and (6) Reverse logistics (Rodrigues Vaz et al., 2017). Even though Mayyas et al. (2012) see a challenge in the fundamental interpretation of the real life cycle span of a vehicle, there exists a consolidated suggestion for the lifetime mileage of cars differentiated by segment (Weymar & Finkbeiner, 2016). Tools that are used to communicate product sustainability impacts are often Environmental Product Declarations (EPD) or similar reports[2] that contain LCA results (ISO 14025, 2006; Mayyas et al., 2012). In the field of product sustainability, the automotive industry at large is following an incremental strategy, meaning that they aim at stepwise improvement of the sustainability impacts of their products (Mayyas et al., 2012;

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[2] Different OEM publish life cycle impacts of their products under different names. For example, Volkswagen AG publishes their results as *Environmental Commendations*, Daimler AG as *Environmental Certificate* and BMW Group as *Environmental Reports.*

Rodrigues Vaz et al., 2017). While being a commendable effort, this might not be enough for OEM to meet the environmental standards calling for radical innovations in the automotive industry (Rodrigues Vaz et al., 2017). The move of OEM towards more revolutionary measures and niche products only follows increased external pressure (like regulations or an external actor) or disruptive influences (like a new technology or a new CEO) (Mazur et al., 2015). For instance, Daimler AG launched the Smart fortwo in 1998, a fuel efficient two-seated urban vehicle. The smart brand, however, had been proposed initially by an external actor - Swatch - as it reflected their vision of future mobility (Mazur et al., 2015). Although the smart brand generated losses for Daimler, it contributed to lowering their average fleet emission from 230g CO_2/km in 1995 to 180g CO_2/km in 2005 (Mazur et al., 2015). At BMW, a new CEO initiated 'project i' which brought forth the BMW i3, a small lightweight battery electric vehicle (BEV) designed for the use within a city (Mazur et al., 2015).

The recent discussion around the gap between automobile emissions as determined in lab conditions and the so-called real driving emissions (RDE) puts more pressure on OEM (Hooftman et al., 2018). This might be accelerating the already started electrification of drive trains because electric vehicles (EV) do not emit local emissions and are also perceived as more ecological (Liu & Meng, 2017). But also from an LCA perspective, EV are often better over their lifecycle compared to conventional vehicles. Karaaslan et al. (2017) showed in a study that compared differently powered sports utility vehicles (SUV) that the Global Warming Potential (GWP) of BEV was lower over the entire life cycle than comparable models with an internal combustion engine (ICE). This was the case even though the impact of the manufacturing of BEV clearly surpassed that of conventional models due to the production of the battery. That means, however, that depending on the production of the vehicle, the mode of operation and especially the way of electricity generation, BEV might not necessarily be preferable to conventional vehicles in terms of GWP over the entire life cycle. Mayyas et al. (2017) showed that in the United States lightweight ICE vehicles performed better over life cycle in terms of GWP than BEV. Therefore, life cycle based product sustainability assessment is needed to ensure that the overall sustainable solution is found (Lehmann et al., 2015, 2017; Minkov et al., 2018; Bach et al., 2018).

1.3 Life Cycle Sustainability Assessment (LCSA)

Life cycle based assessment of products has to be applied in order to fully evaluate product sustainability impacts. For the life cycle based analysis of environmental impacts, Life Cycle Assessment (LCA) is the established method, especially after being standardized by the International Organization for Standardization (ISO) via the norms 14040 and 14044. According to the triple bottom line principle, however, pure LCA is not sufficient as all three sustainability dimensions, i.e. environment, economy and society, have to be considered together. To close this methodical gap, Life Cycle Sustainability Assessment (LCSA) was developed by Walter Klöpffer and Matthias Finkbeiner (Klöpffer, 2003; Finkbeiner et al., 2008; Klöpffer, 2008; Finkbeiner et al., 2010). They suggested a complementary application of three life cycle analyses, each addressing one sustainability dimension. They formalized their approach as follows:

$$LCSA = LCC + LCA + S - LCA \tag{1}$$

LCSA is therefore the complementary application of Life Cycle Costing (LCC) (Swarr et al., 2011), Life Cycle Assessment (LCA) (ISO 14040, 2006; ISO 14044, 2006; Klöpffer & Grahl, 2014) and Social Life Cycle Assessment (S-LCA) (Dreyer et al., 2006; Jørgensen et al., 2008; Benoít Norris & Mazijn, 2009). In the following sub-sections, the three underlying methods are presented individually. The current state of LCSA in the context with the automotive industry is presented in section 1.4.

1.3.1 Life Cycle Assessment (LCA)

LCA is the only one of the three methods that is internationally standardized (Finkbeiner et al., 2006). The procedure and phases of an LCA are depicted in Figure 2.

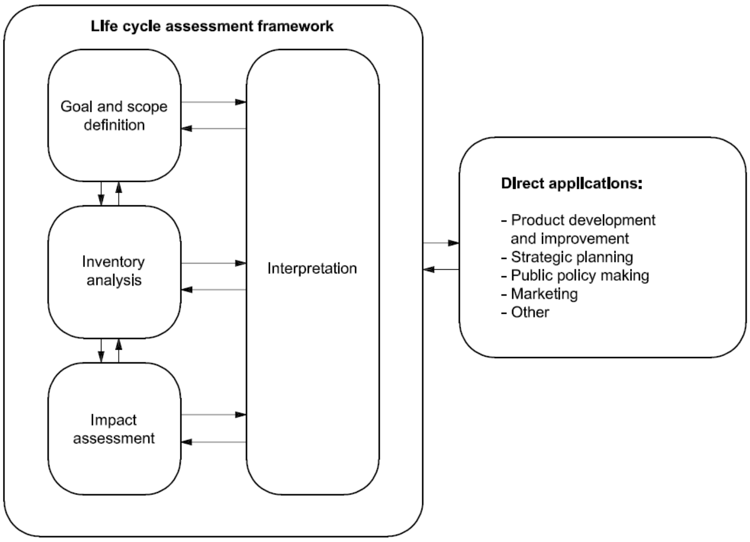

Figure 2: Phases of an LCA (ISO 14040, 2006; ISO 14044, 2006).

An LCA consists of 4 phases. During the **goal and scope definition** the motivation for the study, its intended use and the target audience are defined. The scope defines the extent and depth of the study that are required to reach the intended goal. Furthermore, all crucial parameters are defined in this phase, i.e. the functional unit[3], the system boundaries[4], the allocation method[5] and the impact categories and the impact assessment method, which translates the material flows (e.g. CO_2 emissions) into environmental impacts (e.g. GWP). Furthermore, all assumptions, limitations, critical review points and requirements regarding data quality have to be defined in this first phase. During the **inventory analysis**, the data and material flows for the defined system are collected, validated and referenced to the product and the functional unit, respectively. This phase usually constitutes the majority of labor in an LCA. The **impact assessment** consists of three mandatory and three optional components. The first mandatory step is the selection of impact categories and indicators as well as the characterization model which translates the results of the inventory analysis in life cycle impact category indicators from which an impact category result is calculated. Furthermore, classification[6] and characterization[7] are mandated. The optional components are normalization[8], grouping[9] and weighting[10]. In the **interpretation phase**, the results of the inventory analysis and impact assessment are analyzed. With regards to the specifications and limitations defined in the goal and scope definition phase, conclusions are drawn and courses for further actions are suggested.

In the context of environmental sustainability, the reduction of impacts, or ideally their reversal, is regarded as sustainable or improving sustainability.

Even though, LCA is the most advanced of the three methods and internationally standardized, there still are gaps and limitations. A big issue is still the discussion around allocation rules as the scientific community is not clear on how to correctly solve allocation problems (Freidberg, 2013; Finkbeiner, 2014). Furthermore, the evaluation of impacts on established topics like biodiversity are still unclear, or if and how to consider issues like littering or animal well-being with an LCA (Finkbeiner, 2014). For Klöpffer & Grahl (2014) this issue along with questions around system expansion could be solved by

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[3] The functional unit is the quantified product description which all results of the study are related to.

[4] The system boundaries determine which process belongs to the system that is examined. Ideally, system boundaries are only crossed by elementary flows, i.e. unaltered raw materials that are extracted from nature or unaltered emissions that are released into nature.

[5] If an observed process has more than one product as an output, it has to be determined how the environmental impacts of that process should be allocated to those several products. This is done by applying allocation methods. For example, the breeding of a cow eventually leads to meat and hide as co-products. The lifetime emissions of the cow's upbringing can be distributed, inter alia, by weight of the co-products or their price.

[6] Classification is the assignment of inventory analysis data to the impact assessment category. For example, the methane emissions are assigned to the impact assessment category Global Warming Potential.

[7] Characterization is the term for calculating impact category indicator values from the inventory analysis data according to the respective characterization models. For example, 1 kg of Methane emissions is translated into 25 kg of CO_2e as its Global Warming Potential is 25 times as intense over 100 years (Forster et al., 2007).

[8] Normalization expresses the impact assessment category indicator in relation to a reference point. An example would be to relate the CO_2e emissions of one process to the emission amount on national level.

[9] Grouping is used to group together indicator results and assigning them, for instance, a hierarchical grading like "low", "medium" and "high" priority.

[10] Weighting is applied when several indicators are combined in order to express possible differences in importance of one indicator compared to another.

conventions that the community agrees upon. Further uncertainties pertain to the integration of new impact categories that also look at emissions are not as clearly quantifiable, like "biological emissions" and "destruction of landscape" and whether impacts on the technosphere should be included or not (Klöpffer & Grahl, 2014).

1.3.2 Life Cycle Costing (LCC)

The method of Life Cycle Costing (Swarr et al., 2011) was originally devised to determine the arising costs for all affected parties along the life cycle of a product. Hunkeler et al. (2008) differentiate between three types of LCC — the conventional, the environmental and the societal LCC. The conventional LCC is dealing with the collection of internal costs that are related to the product life cycle and arise to all affected parties — producers, suppliers, consumers and end of life actors. Environmental LCC additionally aims to internalize external costs, i.e. costs that arise due to environmental impacts and are borne by society instead of the consumer or producer of the respective product, that are likely to be internalized in the decision-relevant future, e.g. through taxes. Furthermore, taxes and subsidies are taken into account and even non-monetized LCA results can be included. In addition to that, Societal LCC looks at costs that arise to anyone in society, whether today or in the long-term, by ideally including all externalities associated with the product's life cycle. To carry out LCC complementary to LCA and S-LCA within LCSA, the risk of double counting has to be avoided meaning that only the conventional form of LCC should be chosen or the form of Environmental LCC, where only the internalization of environmental impacts that are expected to be internalized in the decision-relevant future are considered (Finkbeiner et al., 2010; UN, 2011). In addition to the different types, an LCC can also have different points of view. For example, when looking at the life cycle costs of a vehicle, then first the costs of the manufacturer should be looked at followed by the costs of the consumer in the use phase. Additionally, the societal point of view takes costs into account that are connected to the life cycle of a vehicle but not borne by manufacturer or consumer, e.g. infrastructure (Rebitzer et al., 2003). An LCC has the same structure as an LCA but without an impact assessment as there is still uncertainty in how to assess the impacts of the life cycle inventory of an LCC (Gluch & Baumann, 2004; Grießhammer et al., 2007; Swarr et al., 2011; Neugebauer et al., 2016). Improvement potential for LCC still exists in expanding the scope to non-monetary economic aspects (Curran et al., 2004; Neugebauer et al., 2015, 2016). Also, Wood & Hertwich (2013) pointed out that the individual goal of life cycle actors, minimizing costs, stands in contradiction to the societal goal, maximizing value added. It is therefore unclear, how these two competing goals should be evaluated within LCC. Furthermore, there are critical voices that question the usefulness of LCC within LCSA entirely (Jørgensen et al., 2010; Heijungs et al., 2013).

1.3.3 Social Life Cycle Assessment (S-LCA)

According to the UNEP/SETAC Life Cycle Initiative (2011), Social Life Cycle Assessment is used within LCSA to assess impacts on the social dimension. It was also structured similarly to LCA to enable complementary assessments (Ness et al., 2007; Finkbeiner et al., 2010). In 2009, the "Guidelines for Social Life Cycle Assessment of Products" were published by the UNEP/SETAC Life Cycle Initiative that consolidated the state of the art in the field of S-LCA research at the time into one guiding document (Benoít Norris & Mazijn, 2009). For the assessment they propose to combine impact categories that affect the human well-being and possible stakeholders[11] that could be affected by the product in question. To bring these two elements together, the guidelines introduce sub-categories that result from the possible combinations of stakeholders and impact categories. For example, if the stakeholder worker is chosen and combined with the impact category human rights, possible sub-categories could be "child labor" or "fair wage". If the impact category human rights is combined with the stakeholder consumer, the labor related sub-categories do not apply. Other issues might arise, however, like data privacy. Practical guidance and suggestions on which categories and indicators to choose can be found in the "Guidelines for Social Life Cycle Assessment of Products" and the accompanying "Methodological Sheets" that were also published by the UNEP/SETAC Life Cycle Initiative (UNEP/SETAC, 2013).

For S-LCA, there is still the most research needed (Finkbeiner et al., 2010; Lehmann et al., 2013; Klöpffer & Grahl, 2014; Karlewski, 2016). In comparison to LCA, the assessment of social impacts poses additional challenges. Often, there exists no causal relation between a production process and a social impact on stakeholders or it is insufficiently studied (Dreyer et al., 2006; Jørgensen et al., 2008; Hutchins & Sutherland, 2008; Benoít Norris & Mazijn, 2009). Instead, the crucial influence factor is seen to be the conduct of a company as this can more clearly foster or impede social conditions (Dreyer et al., 2006; Jørgensen et al., 2008; Martínez-Blanco et al., 2015). Figure 3 shows the difference in focus between LCA and S-LCA.

Even though S-LCA has the objective to assess the social impacts over the entire life cycle of a product, the way of how to assess impacts in the use phase is still in formation as those are not linked to a company's conduct but are directly connected to the product (Dreyer et al., 2006; Parent et al., 2012). Another challenge compared to LCA is the difference in the type of data that is used in S-LCA. Whereas LCA deals with mostly quantitative data, S-LCA often resorts to semi-quantitative and even qualitative data (Benoít Norris & Mazijn, 2009). Regarding data availability on social risks,

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[11] Stakeholder are people or groups of people that have a vested interest in an activity or decision of a company. In the context of an S-LCA all those are considered as stakeholder that directly or indirectly are involved in or affected by the producing company or its product (e.g. workers, local communities or consumers) (Benoít Norris & Mazijn, 2009).

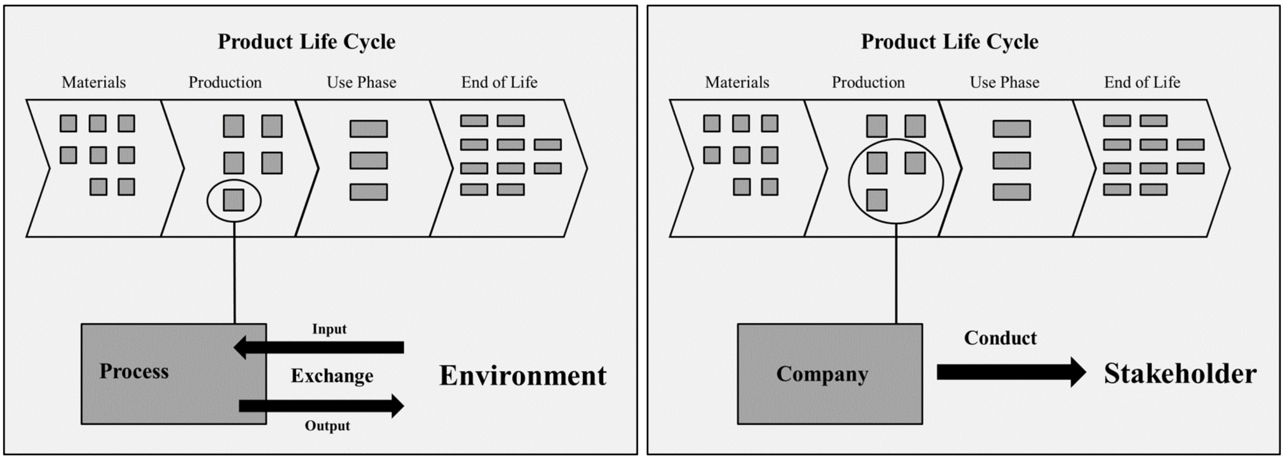

Figure 3: Difference in focus of LCA and S-LCA. LCA focusses on processes for the assessment of environmental impacts (left), whereas S-LCA focusses on company conduct for the assessment of social impacts (right). Adapted from Dreyer et al. (2006).

i.e. the risk for social violations, there exist databases that assess these risk at country level — like Maplecroft (Verisk Maplecroft, 2017) — and even industry sector level — like the Social Hotspot Database (Benoit-Norris et al., 2012). General criticism of S-LCA comes from Iofrida et al. (2018) that see the method development struggling due to a missing epistemological position and grounding in social sciences. They furthermore regard the approach to develop S-LCA in analogy to LCA as inappropriate because the natural science driven LCA might not be an adequate blueprint for the multi-paradigmatic nature of S-LCA. Contrary to their pessimistic appraisal of S-LCA, Traverso (2018) sees the method on a similar route towards harmonization as LCA 50 years ago. Since the publication of the "Guidelines for Social Life Cycle Assessment of Products" in 2009, the field of S-LCA has garnered increased interest. Many case studies have been carried out and several other guiding documents have been published, e.g. the Pré Social Roundtable Handbook of Product Social Impact Assessment (Fontes et al., 2015), the UN Guiding Principles on Business and Human Rights (UN, 2011) and the WBCSD Social Capital Protocol. To gather and consolidate the methodological advancements and practical experiences that have been gained since the publication of the UNEP/SETAC guidelines in 2009, there are efforts underway to prepare an updated guidelines document that aims at gathering the learnings from existing S-LCA case studies, harmonizing the plethora of S-LCA methods and to integrate SOLCA to broaden the scope of the current guidelines.

1.4 Review of LCSA and Potential for its Adoption at an Automotive Company

The methodological and practical state of the art of LCSA and its adoption in the automotive industry have been determined in a publication by Tarne et al. (2017). A structured literature review on the state of the art of LCSA was conducted and combined with the identified needs to operationalize product sustainability assessment within the automotive industry. So far, the LCSA framework has not been fully adopted at a company to assess the sustainability impacts of vehicles in order to support decisions for improvement measures aiming at the overall product sustainability impacts (Tarne et al., 2017). Tarne et al. (2017) identified main challenges for the operationalization of LCSA and the corresponding research needs that have to be addressed to overcome them. Out of the main challenges found and explained in Tarne et al. (2017), the following were chosen to be addressed in this thesis:

- **Indicator selection and primary data collection in S-LCA**

  The plethora of possible social topics that can be addressed and abundance of indicators that can be chosen for the respective social topics is a challenge for putting S-LCA into practice at an automotive company. Additionally, the limited data availability on social impacts poses an impediment to the operationalization of S-LCA and thus LCSA.

  Research Need: A clear procedure to choose social topics consistent with company specific strategies and to derive indicators accordingly. Furthermore, there is a need for a method to conduct social hotspot assessment for an automotive company as this can give a first indication of social risks. This is needed to guide efforts for primary data collection which reduce the efforts to conduct an S-LCA to a feasible level.

  ⇒ This research need is addressed by **Research Objective 1** (c.f. section 1.5).

- **Interpretation of LCSA results for decision makers**

  Clear and understandable presentation and interpretation of LCSA results for non-expert decision makers is a challenge that, if not addressed properly, could impede the application of LCSA results in a company context.

  Research Need: A weighting set that enables the aggregation of sustainability dimensions. Therefore, there is a need to determine what weights decision makers would assign to the individual sustainability criteria or dimensions.

  ⇒ This research need is addressed by **Research Objective 2** (c.f. section 1.5).

- **Implementation of LCSA results in decision process**

  Even if LCSA results can give a clear indication on a preferable option, their implementation in company decision processes, e.g. on product design, is still hindered due to the challenge for decision maker to interpret them alongside known indicators like cost and quality.

  Research Need: A mechanism that enables the implementation of LCSA results in the decision making process alongside established indicators like cost.

  ⇒ This research need is addressed by **Research Objective 3** (c.f. section 1.5).

The publication by Tarne et al. (2017) — "Review of Life Cycle Sustainability Assessment and Potential for its Adoption at an Automotive Company" — is presented on pages 13–36. The goal of this thesis and the definition of the corresponding research objectives to address the outlined impediments to putting LCSA into practice are presented in section 1.5 (p. 36).

sustainability

MDPI

Review

Review of Life Cycle Sustainability Assessment and Potential for Its Adoption at an Automotive Company

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Abstract: The aim of this paper is to guide the next steps of a PhD thesis through a structured review of the state of the art and implementation of Life Cycle Sustainability Assessment (LCSA), and to identify challenges and potentials for its adoption at an automotive company. First, the structured literature review was conducted on LCSA to screen the current methodological and practical implementations and to identify the main research needs in the field. Second, a research on the current status of LCSA within the automotive industry was carried out by means of investigation of published sources of 15 Original Equipment Manufacturers (OEM). By combining the results of both steps and consulting with decision makers, the challenges and potential for adopting LCSA at an automotive company were identified. The main challenges for adoption of LCSA were found to be: (1) the consistent execution of the three life cycle based assessment methods; (2) the comparatively low maturity of Social Life Cycle Assessment (S-LCA); and (3) the adequate presentation and interpretation of results. Next steps towards implementation would be a case study to gather experience on the combined execution of the three life cycle based assessments at an automotive company. Furthermore, it should be determined what the needs of decision makers at an automotive company are regarding the aggregation and interpretation of environmental, social, and economic impacts.

Keywords: LCSA; structured literature review; product sustainability assessment

1. Introduction

Products influence all three dimensions of sustainability, i.e., economy, environment, and society. Their manufacturing and often their usage is linked to resource use, land use, and pollution of the environmental media. Furthermore, products have an effect on both economy and society because their life cycle, i.e., manufacturing, use, and disposal, give rise to employment and contribute to value generation. In addition, products satisfy the needs of customers. When assessing sustainability, how sustainability is understood has a decisive impact on the mode of assessment. There are two fundamentally different approaches to assessing sustainability, the first one is viewing the three dimensions of sustainability partially independent from one another while the second one understands the three dimensions to be integrated and thus not independently manageable [1,2]. Product sustainability assessment should take the impacts over the entire product's life cycle into account [3].

The most accepted framework and life cycle based approach for assessing a product's impacts on all three sustainability dimensions in an integrated way is Life Cycle Sustainability Assessment (LCSA) [3–7]. It is currently regarded as the only viable framework for comprehensive sustainability assessment of products [8,9]. It can be understood as the evaluation of the combined positive and negative effects of a product over its entire life cycle on the three dimensions of sustainability. This comprehensive assessment can be expressed by the following conceptual formula (1):

$$\text{LCSA} = \text{LCA} + \text{S-LCA} + \text{LCC} \tag{1}$$

where LCA (Life Cycle Assessment) denotes the conventional environmental life cycle assessment [10,11], S-LCA (Social Life Cycle Assessment) represents the assessment of positive and negative social impacts along the product life cycle [12,13]., and LCC stands for Life Cycle Costing, the assessment of economic impacts along product life cycle [14,15]. The interpretation of LCC results can express sheer costs or more broadly impacts (e.g., change of wealth of economic well-being of stakeholders). This research adheres to the economic interpretation as the improvement of sustainability in the sense of the Brundtland report [16] is captured by the combined LCSA framework. Thus far, no structured literature review has been conducted on the state of the art of LCSA. Sala et al. [17] is the most complete analysis of the state of the art of the LCSA against ontological, epistemological and methodological aspects of the ongoing scientific debate on sustainability but no structured review of the current published references on LCSA has been made even in this publication.

The automotive industry recognizes the need for improvement when it comes to the sustainability impacts of the transport sector. To efficiently and effectively work on the improvement of sustainability impacts, their quantification is an important step. Adopting LCSA within the automotive industry would enable a comprehensive approach on measuring and managing product sustainability. An overview of the state of the art of LCSA and its implications for adopting it at an automotive company is instrumental for setting the implementation effort off in the right direction. Thus far, no analysis of the state of the art of LCSA in relation to the automotive sector has been carried out.

The aim of this paper is to collect and organize the main references on LCSA in a structured way, and to build a complete picture of the theoretical and practical implementation of this methodology in the automotive sector. Afterwards this picture was used to identify the main needs for further research in the field of LCSA. For guiding specific research efforts towards the implementation, the challenges and potential for LCSA at an automotive company were identified. These results can support further researchers in this field to easily identify further steps needed in the adoption of LCSA at an automotive company. In any case, it represents the first step of a PhD thesis on LCSA at an automotive company. The necessity to draw a structured review that is easy to follow, generic and specific for the automotive sector is led by the need to communicate the results to experts and non-experts of the sustainability field. The PhD thesis has been funded by the BMW Group in cooperation with TU Berlin which is the first time a research on the operationalization of LCSA has been founded by an automotive company.

The research was divided into three steps:

(A) Structured literature review on LCSA to identify the main research needs.
(B) Identification of currently communicated life cycle based product sustainability assessment of automotive companies to establish a benchmark.
(C) Mirroring of retrieved results to decision makers and identification of challenges and potential of LCSA at an automotive company.

2. Materials and Methods

The first step consisted of a structured literature review on the state of the art of LCSA in general (cf. Step A in Section 1). Here, only literature that assessed at least two of the three sustainability pillars along the product life cycle was considered. In considering those literature references, differences were made according to methodologies for interpreting the LCSA results, e.g., Multi-Criteria Decision Analysis (MCDA). This does not mean that in the review an exhaustive state of the art of MCDA as well but instead that only those references where the MCDA has been applied to interpret LCSA results were included and discussed. The second step turned toward the automotive industry and identified the status of LCSA at the top Original Equipment Manufacturers (OEM) in the world (cf. Step B in

Section 1). By taking the results of both steps into account and consulting with decision makers, the challenges and potential for operationalization of LCSA at an automotive company were derived (cf. Step C in Section 1).

2.1. Structured Literature Review of LCSA Research Field

For the first step of this research, a structured literature review approach was chosen to determine the state of the art of LCSA. Structured literature reviews provide collective insight into a field and enable researchers to develop a reliable knowledge base [18]. As it is a structured method, it reduces researcher bias and, ideally, yields reproducible results [19].

The steps of the literature review were as follows [18–22]:

(1) Formulation of review protocol
(2) Formulation of research question
(3) Formulation of inclusion and exclusion criteria
(4) Search for literature
(5) Evaluation of literature

2.1.1. Review Protocol

A protocol has to be set up that documents the undertaken steps and criteria [18,23]. Table 1 shows the review protocol of the conducted literature review.

Table 1. Review protocol.

Review question	"What are the thematic fields of research regarding LCSA and what are the identified needs for future research?"
Inclusion criteria	*Title*: "Life Cycle Sustainability Assessment" or more than one sustainability dimension
	Abstract and full text: at least two sustainability dimensions and when it dealt with the method
Exclusion criteria	Book reviews; non-English publications; addressing of just one sustainability dimension
Literature search	*Sources*: specialist, scientific online platforms (ScienceDirect, WileyOnline, SpringerLink, MDPI), citations in identified literature
	Search phrase: "Life Cycle Sustainability Assessment" in title and abstract with no limitation to publication year
Evaluation	Following information were extracted from the publication and transferred into an excel document: specifications of publication, scope of publication, results of publication

2.1.2. Research Question

The following research question guided the structured literature review: "What are the thematic fields of research regarding LCSA and what are the identified needs for future research?" In this first part no particular reference to the automotive sector has been made to catch and identify all scientific references related to the LCSA. For the same reason mentioned above, the research questions have been formulate quite generic and wide to collect as much inputs as possible.

2.1.3. Inclusion and Exclusion Criteria

(A) Inclusion Criteria

a. *Title*: When the phrase "Life Cycle Sustainability Assessment" was found; when more than one dimension of sustainability was addressed; when dimensions were not clear but "sustainability" was mentioned.

b. *Abstract:* When at least two sustainability dimensions were addressed and when it applied, evaluated, developed, or advanced a method.

c. *Full text:* When at least two sustainability dimensions were addressed and when it applied, evaluated, developed, or advanced a method.

(B) Exclusion Criteria

a. Book reviews
b. Non-English publications
c. Addressing of just one sustainability dimension

The identified literature was then analyzed in regards to the main topics that they dealt with and were grouped by the authors according to major overarching thematic fields. The thematic fields were derived by the authors and were guided by the topics that emerged as well as the motivation to limit the number of fields. As the aim was to generate a broad overview that is also comprehensible for non-experts of LCSA it was decided to not differentiate between too many thematic fields.

2.1.4. Search for literature

To define the search for literature, the sources (A) and keywords (B) had to be specified.

(A) Sources

First, the sources for the literature search were defined. As several approaches to identifying relevant literature are recommended [18,20], a specialist was asked about the main sources for the respective question [21]. This yielded the *International Journal of Life Cycle Assessment*, the *Journal of Cleaner Production*, the *Journal of Industrial Ecology* and the open access journal *Sustainability* as the most relevant sources.

This seemed reasonable as the literature review of Mattioda et al. [24] on S-LCA also found those four journals to contain the majority (68%) of the relevant publications. To extend the scope, the search was conducted on the scientific portals that host the respective journals, i.e., SpringerLink (*International Journal of Life Cycle Assessment*), WileyOnline (*Journal of Industrial Ecology*), ScienceDirect (*Journal of Cleaner Production*), and MDPI (*Sustainability*).

Furthermore, cross checking and searching of citations in the identified literature was applied in this research to identify relevant publications not included in the initial sources [20,25].

(B) Keywords

Initially, the keywords "life cycle", "sustainability", and "assessment" were defined as Life Cycle Sustainability Assessment and its status in the automotive industry was chosen as the object of research. As the search with this set of keywords yielded more than 10,000 results, a more stringent search term was defined to achieve a more manageable amount of hits. However, even partially limiting the search term by searching for instance for "life cycle" and "sustainability assessment" yielded hits way in excess of 10,000. Eventually, the exact phrase was therefore limited to "Life Cycle Sustainability Assessment" which was searched for in titles and abstracts with no restriction to publication year. The search was done in July 2015. This resulted in 274 initial hits. Figure 1 shows the process of the literature search and subsequent identification of relevant publications.

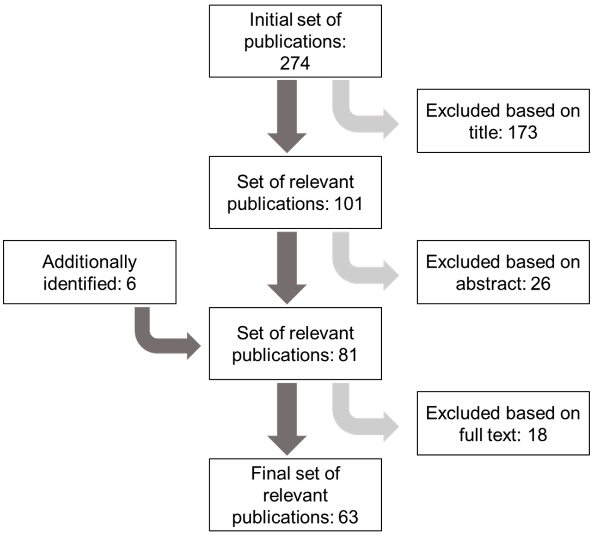

Figure 1. Schematic illustration of the literature search and selection process.

According to Miemczyk et al. [26], the titles of the identified publications were then checked against the inclusion/exclusion criteria (cf. Table 1), which diminished the pool to 101 relevant items. In the next step, the abstracts were evaluated regarding whether they fit the inclusion/exclusion criteria. This resulted in a total of 81 relevant publications for the evaluation. After full-text analysis, another 18 publications were excluded, leaving a set of 63 relevant publications for the state of the art analysis.

2.1.5. Evaluation

Following the example of Zimmer et al. [25], classification categories were defined for a systematic evaluation of the identified body of relevant literature.

(A) Specifications of Publication

The first classification category serves the purpose of capturing the basic parameters of the publication. Thus, year of publication, type of publication (journal article, conference report, book chapter, etc.), and name of source were chosen for this category.

(B) Scope of Publication

In order to categorize the publications pertaining more to their content and scope, the following parameters were chosen for this category:

- **Type of study (empirical, theoretical, or both)**

  Following Wacker [27] who divide papers into "analytical" and "empirical", the types of study were divided into "theoretical", meaning the development of a method or framework without testing it or structuring/summarizing of existing approaches, indicators, views, etc. "Empirical" denotes a publication that employs a case study without seeking major adjustments to the method applied. The classification "both" was used for publications that either developed a new method or advanced an existing one and tested this method with a case study.

- **Industry sector**

In the case a publication or case study was directed at, or carried out in, a specific industry sector, this was recorded here.

- **Addressed sustainability dimension**

  It was recorded which sustainability dimensions were addressed in a publication. All studies that addressed just one dimension were excluded from the research.

- **Type of case study**

  Case studies were categorized into "full case study" or "numerical example" depending on whether they collected real data or whether they tested a method using fictional values.

2.2. Evaluation of Current Status in the Automotive Industry

The second step of the research was conducted independently of the structured literature review on LCSA. As the initial search for papers on LCSA in the automotive industry basically just yielded the paper by Traverso et al. [28], the publications (mainly sustainability reports and product related information) of the 15 largest automobile manufacturers by production volume were analyzed in order to get an overview of the current status of Life Cycle Sustainability Assessment in the automotive industry.

2.3. Implications for Application at an Automotive Company

In order to identify the most pressing challenges out of the ones identified by the two previous research steps (Sections 2.1 and 2.2) for the adoption of LCSA, decision makers at an automotive company were consulted. The consulting process was carried out with 12 decision makers who were at least at the level of department head. They were first presented with the findings of the two previous research steps and were asked in a semi-structured interview whether they agreed with the identified challenges and potentials for adoption of LCSA within an automotive company. In a second step, they were asked about their expectations towards an operational LCSA framework. The interviews were conducted in person.

3. Results and Discussion

3.1. State of the Art of LCSA

The drastic reduction in initial hits from way over 10,000 to 274 after limiting the literature search string indicates a high "noise", meaning not pertinent literature, when searching for publications related to life cycle sustainability assessments. This is not surprising as the individual key words tap into separate extensive fields of research which is why the search had to be limited in the first place. The comparatively low amount of initial hits with the limited search string indicates that the specific field of Life Cycle Sustainability Assessment is still in development. This indication is substantiated by the further findings in this research. The final set of analyzed publications of 63 compares to other structured literature reviews [20,26,29].

A detailed overview of the analyzed publications and their classification along with a list of the initial hits can be found in the supplementary material to this article.

3.1.1. Bibliometric Results

In the following, bibliometric evaluations of the set of examined publications are presented visualizing some characteristics and developments in the field of LCSA research. Figure 2 shows the distribution of publications by year.

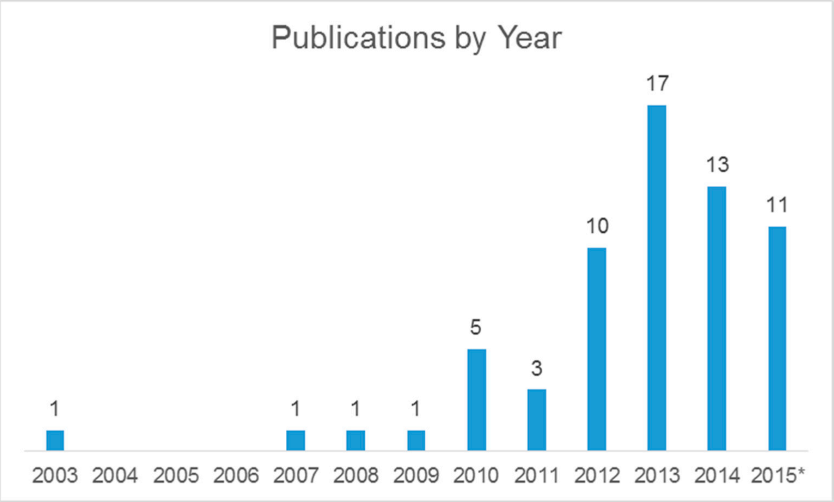

Figure 2. Distribution of publications by year. * The review was conducted in July 2015, thus not all publications in 2015 could be considered in this research.

It can be clearly seen that the majority of publications appeared after 2011. An increasing trend can be assumed as the figure for 2015 only includes publications up until July 2015 when the data for this research were gathered. The fact that most of the LCSA related research has been published after 2011 when the UNEP/SETAC guidelines were published re-affirmed the need for this research. The peak in publications in 2013 can be explained be the special issue on Life Cycle Sustainability Assessment by *The International Journal of Life Cycle Assessment* out of which eight papers were included in this review. Table 2 displays the publications by source.

Table 2. Distribution of publication by source/journal. All sources that contributed just one publication were subsumed under "Rest".

Name of Journal	No. of Publications
The International Journal of LCA	21
Sustainability	10
Journal of Cleaner Production	5
Journal of Industrial Ecology	4
Applied Energy	2
Journal of Remanufacturing	2
Procedia CIRP	2
Rest	17
Total	**63**

It is evident that *The International Journal of Life Cycle Assessment* is the dominating medium in the field of LCSA, followed by the online journal *Sustainability*. The rather large share of sources under "Rest" indicates that the chosen research method enabled the identification of relevant publications that were not contained in the initial sources of the search thus proving the viability of the chosen method. Figure 3 shows the characteristics of the examined set of publications in regards to the type of study.

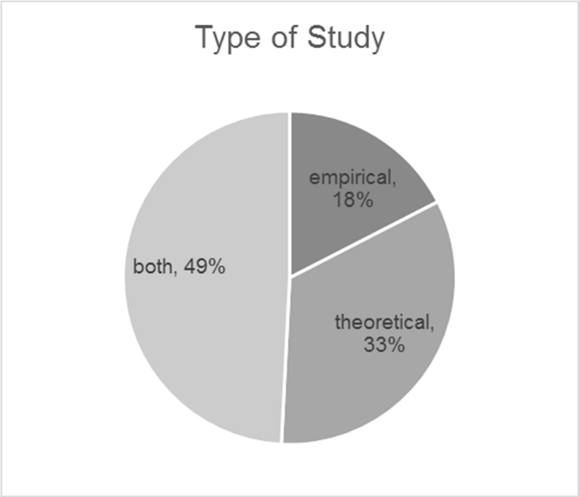

Figure 3. Publications by type of study.

As can be seen in Figure 3, 52 of the examined publications had theoretical elements (mostly development of assessment methods), whereas 42 had empirical elements (mostly case studies). As the group of studies that combine empirical and theoretical elements is mainly comprised of method developments that are subsequently tested by case studies it can be summarized that the field of analyzed literature leans more strongly towards method development rather than gathering empirical evidence using existing methods and frameworks. This is underpinned by only 11 publications containing pure case studies. This distribution of theoretical versus empirical studies shows characteristics of a nascent to intermediate state regarding the maturity of the field of research [30]. When looking at the numbers of sustainability dimensions addressed, 80% of publications addressed all three dimensions while 20% only addressed two sustainability dimensions. Publications that addressed just one dimension were excluded from this research and were thus not present in the review.

The distribution of industry sectors that were dealt with in the examined publications is presented in Table 3.

Table 3. Distribution of industry sectors that were addressed in the examined publications. Twenty publications did not explicitly address a specific industry sector.

Industry Sector	No. of Publications
-	20
Building	8
Automotive	7
Energy	6
Fuels	5
Waste Management	4
Agriculture	3
Electronics	2
Other	8
Total	**63**

Out of the analyzed publications, 20 did not specifically address an industry sector (see Table 3). That makes for a consistent picture of the examined set as 21 publications used only a theoretical approach (cf. Figure 3) not requiring a link into practice. The most prominent industry sectors were building, automotive, energy, and waste management. These findings underpin that the automotive industry is in the focus when it comes to the sustainability of products. When looking at the countries of origin of the publications, the field was led by Germany (21%) followed by Italy (16%), USA (13%) and the UK (11%).

3.1.2. Classification of Literature by Thematic Fields

The thematic clustering of the research field was not straightforward as the set of examined publications posed a heterogeneous group of publications with no clear "mainstream" discernible or clear delineations between different research foci. This in itself already constitutes a major finding of this review. Nonetheless, it was found that the field could be clustered into four thematic fields: "Framework", "Method Integration", "Case Studies", and "Alternative Assessment Methods". The thematic field "Framework" comprised all research papers that dealt with the principle logic and further development of the framework LCSA rather than singular methods. Papers in the field of "Method Integration" were dealing with the integration of methods, meaning that parts or elements of the higher-level framework were extended, changed, or added. The thematic field "Case Studies" was made up by publications that mainly carried out case studies for LCSA. The last field "Alternative Assessment Methods" denotes the collection of papers that were developing their own assessment method by either incorporating parts of or totally independent from the LCSA framework. Table 4 shows an overview of the thematic fields and the respective publications.

Table 4. Overview of thematic fields of LCSA literature and the according authors.

Thematic Field	Subcluster	Authors
Framework		Klöpffer (2003), Klöpffer (2008), Heijungs et al. (2009), Finkbeiner et al. (2010), Heijungs et al. (2010), Guinée et al. (2011), UNEP (2011), Zamagni (2012), Cinelli et al. (2013), Jørgensen et al. (2013), Sala et al. (2013), Sala et al. (2013), Zamagni et al. (2013), Hoogmartens et al. (2014), Klöpffer and Grahl (2014), Keller et al. (2015), Neugebauer et al. (2015), Traverso et al. (2015)
Method Integration	*LCSA Steps*	Foolmaun and Ramjeawon (2013), Foolmaun and Ramjeeawon (2013), Hu et al. (2013),Pesonen and Horn (2013), Vinyes et al. (2013), Stefanova et al. (2014), Souza et al. (2015)
	Multi-Criteria Decision Analysis	Halog and Manik (2011), Ghadimi et al. (2012), Manzardo et al. (2012), Bachmann (2013), Ostermeyer et al. (2013), Wang and Chan (2013), Buchert et al. (2015), Ren et al. (2015)
	Multi-Regional Input-Output Analysis	Kucukvar and Tatari (2013), Kucukvar et al. (2014), Onat et al. (2014), Onat et al. (2014)
Case Studies		Brandão et al. (2010), Schau et al. (2012), Traverso et al. (2012a), Traverso et al. (2012b), Valdivia et al. (2013), Lu et al. (2014), Martínez-Blanco et al. (2014), Minne and Crittenden (2015), Yu and Halog (2015)
Alternative Assessment Methods		Zhou et al. (2007), Azapagic and Stichnothe (2010), Moriizumi et al. (2010), Jeswani and Azapagic (2012), Nzila et al. (2012), Schulz et al. (2012), Stamford and Azapagic (2012), Luthe et al. (2013), Tugnoli et al. (2013), Ingwersen et al. (2014), Mjörnell et al. (2014), Shuaib et al. (2014), Stamford and Azapagic (2014), Torquati et al. (2014), Dewulf et al. (2015), Hirschberg and Burgherr (2015), Li et al. (2015)

There are different understandings of how to approach product sustainability in general as becomes evident in the different understandings on how to develop a framework, and to Life Cycle Sustainability Assessment in particular as there were plethora of issues addressed and even more alternative assessment methods developed.

Framework

Publications sorted into this category dealt with the principle logic and further development of the framework LCSA rather than singular methods. In this review paper, the acronym LCSA stands for the Life Cycle Sustainability **Assessment** framework as defined by Klöpffer [4,5] and Finkbeiner et al. [6].

The foundations for the LCSA framework as a complementary application of the life cycle assessment methods LCA, S-LCA, and LCC were laid by Klöpffer and Finkbeiner. They gave two options of how this could be achieved: either by carrying out those three separate assessments with the same system boundaries or applying them as three different impact assessments to the same life cycle inventory [5]. In the quasi-guidelines by UNEP/SETAC [3], it was added that, in addition to the same system boundaries of all three methods within LCSA, the same temporal horizons are also to be considered.

The first overview of the status quo of the LCSA techniques LCA, LCC, and S-LCA were given by Finkbeiner et al. [3,7]. They also adopted Maslow's pyramid of needs to reflect the hierarchy of sustainability assessment approaches showing LCSA on top and thus as the most sophisticated product sustainability assessment available. Furthermore, they introduced two presentation tools for LCSA results: the Life Cycle Sustainability Triangle and the Life Cycle Sustainability Dashboard [7,31,32]. Zamagni [9] affirmed that LCSA is the state of the art of life cycle based sustainability assessment.

Sala et al. [17,33] developed a framework in order to structure the field of sustainability science according to the scientific and social paradigm. Then they conducted a meta-review of papers on sustainability assessments in order to check whether they could be integrated in the freshly developed sustainability science framework. In their evaluation, they compared the LCSA framework by Klöpffer and the Life Cycle Sustainability **Analysis** framework by Guinée and colleagues against their sustainability assessment. Both frameworks were considered potentially fit to work as sustainability assessment method, but the framework of Guinée et al. was seen as further along although it still missed translation into operation [17,33]. Therefore, their framework was not further considered for the following steps of this research.

Hoogmartens et al. [34] included Cost-Benefit-Analysis (CBA) in their status report on LCSA as they compared and discussed LCA, LCC, S-LCA, and CBA in regards to the assessment of sustainability resulting in an illustrative framework to show differences and interactions. Environmental LCC was regarded as "pure LCC", full environmental LCC as LCC with monetization of environmental impacts.

In 2014, Klöpffer and Grahl noted that no substantial development on LCSA had taken place and that LCSA could be realized by taking several routes [35]:

(1) LCC + S-LCA as additional impact categories for a life cycle inventory (LCI) of an LCA
(2) Eco-efficiency + S-LCA
(3) LCA + socioeconomic analysis

However, there also exists doubt that a combination of LCA, S-LCA, and LCC would lead to a more comprehensive sustainability assessment. Jørgensen et al. [36] analyzed whether the proposed LCSA could actually assess the effects on sustainability as defined in the Brundtland report. They found that LCA and S-LCA are necessary (even though further development towards assessment of poverty is needed) but LCC is not necessarily an integral part [36].

An extension to the LCSA framework was introduced by Keller et al. [37]. In order to take barriers to implementation into account, thus providing a better basis for decision support, they proposed four elements of evaluation: technology, environment, economy, and social, with only the environmental indicators derived from a clearly life cycle based tool (LCA).

Aiming at facilitating the application of LCSA, Neugebauer et al. [38] proposed a tiered structure to LCSA, depending on the level of barriers to operationalization. Tier 1 provided a set of midpoint indicators that were rather easy to assess and interpret, whereas Tier 3 suggested a comprehensive approach with an extended list of indicators and impact categories calling for a higher effort in execution but yielded a more detailed result [38].

As main future research needs the publications mainly identified the challenges that come with combining three method into one framework consistently [9]. Firstly, the methods displayed different degrees of maturity with especially S-LCA having the challenge of affluence of indicators and difficulty of quantification and relation to product [7,35]. Secondly, the consistency requirement calling for same system boundaries for all methods were considered to demand some further work as it was not clear whether this was always feasible or even conceptually correct [9]. Thirdly, the weighting and aggregation within LCSA should be addressed as this would enable researchers/users to arrive at a combined result more clearly able to support decision makers [7].

Other prominent fields for future research were the communication of LCSA results [37,39,40] and data availability [7,37].

Method Integration

Publications of this category dealt with the integration of methods, meaning that parts or elements of the higher-level framework were extended, changed, or added. The field of method integration could be further divided into publications focusing on **LCSA Steps**, the application of **Multi-Criteria Decision Analysis**, and the application of **Multi-Regional Input-Output Analysis**.

LCSA Steps

Further structuring of the goal and scope phase of LCSA in order to make it more operational is suggested by Hu et al. [41] and Stefanova et al. [42]. This could be achieved by either introducing questions which answers will lead to goal and scope definition [41] or by defining the phase top-down. The latter was suggested by first defining macro-goals followed by a technology map that comprises all possible routes to achieve those goals. Finally, the establishment of a context determines what routes can be taken and what routes have to be excluded, thus setting the scope of the assessment [42].

Following the idea of simplification, the introduction of a Strengths–Weaknesses–Opportunities–Threats (SWOT) matrix as a streamlined LCSA approach that incorporates all three dimensions in a concise manner that is easily to communicate was examined by Pesonen and Horn [43].

Turning to the impact assessment phase, several researchers proposed a simplified impact assessment when comparing alternative scenarios by setting the best performing option in a respective category as 100% and putting the other options in relation [44–46]. The missing consensus on impact pathways motivated Souza et al. [47] to develop a method to define impact categories and impact pathways for the social and economic dimension of LCSA by stakeholder engagement. They derived causal maps from the subjective feedback of stakeholders which resulted in familiar impact categories but also new ones, especially for social: digital inclusion was additionally identified, whereas child labor and collective bargaining were not considered relevant [47].

Future research needs identified were the weighting and measuring of social impacts along with the definition impact categories in economic dimension [44,45], and the conduction of further case studies [41,47].

Multi-Criteria Decision Analysis

Several authors chose to integrate multi-criteria decision analysis (MCDA) in order to be able to combine different indicators into one aggregated result, deriving weights for the respective indicators, and even translate qualitative information into quantitative metrics [48–52]. In order to determine weights, researchers usually applied fuzzy evaluation methods, the Analytical Hierarchy Process (AHP) or a combination of those to translate qualitative evaluations of experts into metrics [48–51,53]. Additional methods like TOPSIS or VIKOR were then used to either determine distance to target [50] or the best alternative [51]. Ostermeyer et al. [54] dealt with the multi-criteria approach using Pareto optimization together with the application of LCC + LCA and defined more than 700 scenarios out of which the preferred option was identified using the Pareto optimal approach. Bachmann [55] derived suggestions for developing LCSA by comparing the external costs assessment and the MCDA within the NEEDS project concluding that both overlap except regarding social indicators and a consequential approach would be preferable when modeling LCSA.

Multi-criteria decision models were also used for a method that was developed to consider sustainability aspects early in the design process by integrating LCSA, engineering design processes, and multi-criteria assessment. The combination resulted in a decision-tree that depicted several design solutions and their impacts which was then analyzed using multi-criteria assessment in order to determine the preferable option [52].

In the field of multi-criteria decision models within LCSA, the definition and choice of suitable criteria in LCC, LCA, and S-LCA was desired for future research [48,52,55]. Furthermore, the integration of dynamic relations between evaluation criteria and other dynamic influences like energy mixes, costs, and discount rates should be looked at [50,54].

Multi-Regional Input-Output Analysis

Several publications made use of extended multi-regional input-output (MRIO) analyses to determine the sustainability impacts of the US building sector [56,57] and different automotive drive-trains [58]. The initial method was developed by Kucukvar and Tatari [56] who used supply and use tables from the U.S. Bureau of Economic Analysis to model supply relationships between industry sectors and therefore simulate a supply chain. Kucukvar et al. [59] expanded the method by Kucukvar and Tatari [56] by using a multi-criteria decision model to determine the ideal alternative under given weights for sustainability dimensions.

Areas for improvement by future research would be a better disaggregation of construction sector in input-output models [56] as well as further development of MRIO tables regarding the granularity of the models and development of dynamic models [57,58].

Alternative Assessment Methods

A considerable proportion of the examined publications were developing their own assessment method either incorporating parts of or totally independent from the LCSA framework. Some authors took LCSA elements and either developed a simplified tool that allows quick assessment of wastewater treatment options based on LCA and LCC [60] or combined LCA and economic indicators to assess biogas production options in Kenya [61]. Other partial integration was done by Luthe et al. [62] who developed a tool that integrates LCA, S-LCA, and economical aspects into product design, while Mjörnell et al. [63] claimed to integrate all three LCSA components into their sustainability assessment, even though they chose indicators from the S2020 Knowledge Matrix instead of established S-LCA sources.

Most of the publications, however, developed their own assessment method, totally independent from the LCSA framework. A rather elaborate assessment method was introduced by Stamford and Azapagic [64] who used 43 indicators grouped into sustainability sections (techno-economic, environmental, social) to assess possible future energy mixes in the UK. They followed up their assessment with a revised method, employing 16 indicators that followed LCA for environmental impacts but own deliberations regarding economic and social impacts to identify the most sustainable energy mix for the UK in 2070 [65]. Similarly, the assessment of primary energy carriers was the focus of the Integrated Sustainability Assessment Framework (ISAF) in which 15 indicators for four sustainability dimensions (adding "technology" to the three conventional ones) were suggested [66]. Energy sources were also assessed by Hirschberg and Burgherr [67] who based their indicators on the NEEDS framework. They resorted to LCA for the environmental dimension, expert judgment for the social dimension, for which the indicators health and security were chosen and capital investment plus fuel costs for the economic dimension. Aggregation of results was achieved by calculation of external costs or through MCDA [67]. In contrast, Moriizumi et al. [68] used a life cycle based assessment of different mangrove management options using just one indicator per sustainability dimension and, similarly, Zhou et al. [69] conducted their assessment based on four indicators that covered environmental and economic impacts.

Two publications suggested the inclusion of already existing methods and indices. Ingwersen et al. [70] proposed to include integrated metrics, such as Ecological Footprint, Green Net Value Added, Fisher Information into sustainability assessment as they can address additional aspects such as carrying capacity (Ecological Footprint) and were already established. Li et al. [71] on the other hand developed a sustainability assessment model by combining LCA, full cost accounting (FCA), health risk assessment, and AHP. Another two publications combined existing LCA studies on biofuels with information from different sources on investment and operation costs for different biofuel plants. A qualitative appraisal of social impacts was done additionally to cover all three dimensions. No combined presentation of results was given [72,73].

The main research needs in this rather diverse set of publications were identified to be more testing of the developed methods [63,64,66], refinement and ranking of indicators [61,70], and the incorporation of input-output analysis [60].

Case Studies

The last of the four identified main fields in the analyzed set of publications was composed of case studies that were applying LCSA methods in order to test their applicability and gain more insights into the implications of the results gained by applying those methods.

The most straightforward application of LCSA was undertaken by Traverso et al. [31,32], Schau et al. [74], and Valdivia et al. [40] who applied the framework to floor coverings, photovoltaic modules, remanufacturing of automobile alternators, and several types of marble, respectively. Traverso et al. [31] introduced the Dashboard of Sustainability as communication tool for assessment results. Furthermore, LCSA was applied by Lu et al. [75] to assess two recycling routes (material recovery or component reuse) of mobile phones in the formal and informal recycling sector in China and by Yu and Halog [76] who assessed a PV array in Australia using LCSA but using a rather basic qualitative approach for S-LCA. They interpreted secondary data (reports, websites, and literature) from main supply chain actors in a color-coded matrix.

A case study using LCSA framework with special focus on S-LCA on mineral and compost fertilizers was conducted by Martínez-Blanco et al. [77] who used data from LCA databases (environmental impacts), LCC (purchase price), and S-LCA (Social Hotspot Database) to assess the sustainability impacts of fertilizers. Two other case studies used a slimmed down approach just applying LCA and LCC to assess flooring options [78] and the effects of different crops on land use [79].

The main research needs identified by the case study publications were the way of weighting between sustainability dimensions and the integration of all three dimensions, e.g., how to deal with inconsistencies in system boundaries [74,75,77]. Further case studies [40,76,77], and improving data availability, especially for S-LCA [74,77], were additionally identified as potential areas for improvement.

3.1.3. Methods Not Identified by the Structured Literature Review

It is worth noting that prominent assessment methods such as eco-efficiency analysis [80], SEEbalance [81,82] or PROSA [83] did not make it into this review, even though they assess two (eco-efficiency analysis) or three (SEEbalance and PROSA) sustainability dimensions. This is a result of the strict search string formulation when the literature search was conducted where no paper on these methods surfaced. This might also indicate another characteristic of the discovered field of papers. As methods from authors with affiliations to companies are hardly present in the analyzed set of publications, it seems that the search for LCSA brought predominantly academic research to light carrying with it a certain view and intended audience. How far this holds true might be an interesting question for further research.

3.2. Current Status in the Automotive Industry

Turning to the results of the second step of this research, only Traverso et al. [28] reported in more detail on the integration of LCSA at the BMW Group. While LCA is integrated into product development process and used to identify and quantify measures for the improvement of environmental impacts there, work is still to be done on integrating social and economic impact assessment. First steps towards the operationalization of Social LCA have been taken by co-founding the Roundtable for Product Social Metrics, an industry-led initiative that produced a handbook on implementing product social impact assessment [84].

The results of the investigation of other sources regarding the current status of LCSA within the automotive industry are presented in Table 5 where all automobile manufacturers are listed that produced more than one million cars in 2014 [85]. Together, they manufactured more than 85% of the world's production volume of cars.

It is evident that virtually all of the top 15 car manufacturers employ LCA as life cycle based sustainability assessment. Only two Chinese manufacturers do not seem to apply LCA. Nine of the manufacturers that employ LCA even had their results certified from a third party to use it in their communication. Noteworthy is the clear focus on "just" LCA, meaning the sole addressing of the environmental dimension of product sustainability. An exception is Ford where all three dimensions of sustainability are addressed by additionally assessing the total cost of ownership as well as a few indicators related to the social impact of the use phase of their product. However, the LCA method applied at Ford still adheres to the ISO standard as formulated in 1997. Furthermore, there are singular efforts to address other sustainability dimensions at other OEMs. Research has been carried out at Volkswagen on LCC [14] and a dissertation on S-LCA at Daimler [86].

When looking at how many of the OEMs communicate their results externally, the number shrinks to roughly half. Those who communicated their LCA results publicly focus mainly on greenhouse gas emissions (GHG) while the presentation of the results was done in comparison with the respective predecessor model or a conventional model when assessing alternative drive systems. Another way was to report the absolute results of the indicators.

It can be summarized that (environmental) LCA is well accepted and practiced throughout the world's top car manufacturers, whereas comprehensive Life Cycle Sustainability Assessment of cars cannot be found. Even though Ford is reporting on social and economic criteria in their Product Sustainability Index, they focus rather on the use phase, thus not taking the entire life cycle into account.

As the automotive industry has to prove to legislators that their products do not contain legally banned substances and that they fulfill the required recycling quotas, they have to determine the material balance of their cars [87]. To efficiently manage the information on material composition of the many components that are sourced externally and thus are supplied by several suppliers to several car manufacturers, the International Material Data System (IMDS) was established in which all this information is entered by suppliers and can be extracted from OEMs.

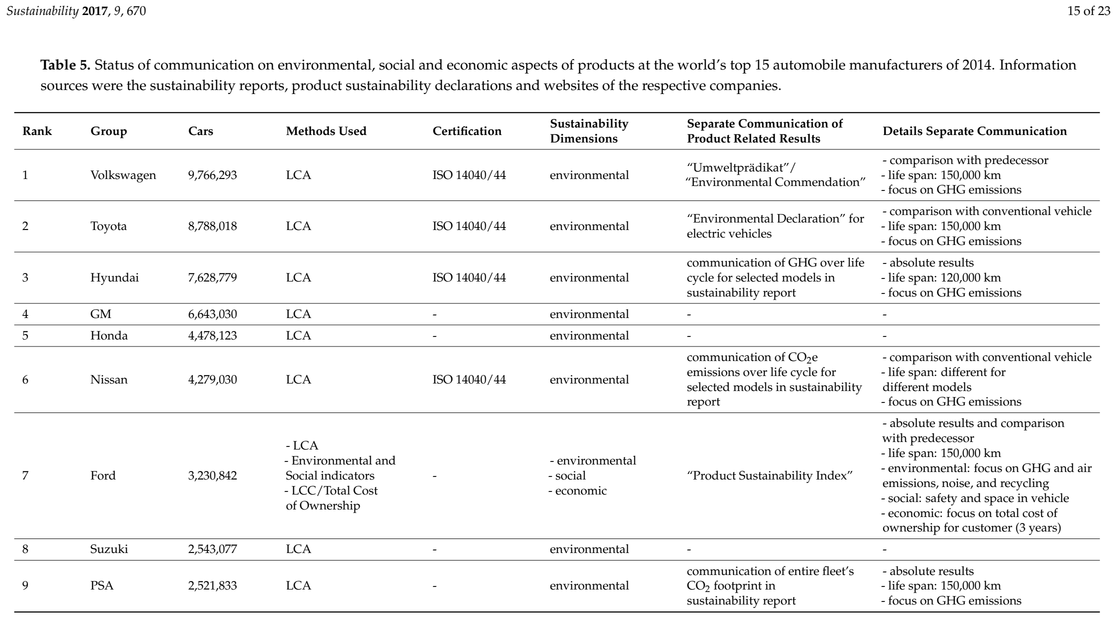

Table 5. Status of communication on environmental, social and economic aspects of products at the world's top 15 automobile manufacturers of 2014. Information sources were the sustainability reports, product sustainability declarations and websites of the respective companies.

Rank	Group	Cars	Methods Used	Certification	Sustainability Dimensions	Separate Communication of Product Related Results	Details Separate Communication
1	Volkswagen	9,766,293	LCA	ISO 14040/44	environmental	"Umweltprädikat"/ "Environmental Commendation"	- comparison with predecessor - life span: 150,000 km - focus on GHG emissions
2	Toyota	8,788,018	LCA	ISO 14040/44	environmental	"Environmental Declaration" for electric vehicles	- comparison with conventional vehicle - life span: 150,000 km - focus on GHG emissions
3	Hyundai	7,628,779	LCA	ISO 14040/44	environmental	communication of GHG over life cycle for selected models in sustainability report	- absolute results - life span: 120,000 km - focus on GHG emissions
4	GM	6,643,030	LCA	-	environmental	-	-
5	Honda	4,478,123	LCA	-	environmental	-	-
6	Nissan	4,279,030	LCA	ISO 14040/44	environmental	communication of CO_2e emissions over life cycle for selected models in sustainability report	- comparison with conventional vehicle - life span: different for different models - focus on GHG emissions
7	Ford	3,230,842	- LCA - Environmental and Social indicators - LCC/Total Cost of Ownership	-	- environmental - social - economic	"Product Sustainability Index"	- absolute results and comparison with predecessor - life span: 150,000 km - environmental: focus on GHG and air emissions, noise, and recycling - social: safety and space in vehicle - economic: focus on total cost of ownership for customer (3 years)
8	Suzuki	2,543,077	LCA	-	environmental	-	-
9	PSA	2,521,833	LCA	-	environmental	communication of entire fleet's CO_2 footprint in sustainability report	- absolute results - life span: 150,000 km - focus on GHG emissions

Table 5. *Cont.*

Rank	Group	Cars	Methods Used	Certification	Sustainability Dimensions	Separate Communication of Product Related Results	Details Separate Communication
10	Renault	2,398,555	LCA	-	environmental	communication of LCA results for electric vehicles	- absolute results and comparison with conventional vehicle - life span: 150,000 km - focus on CML2001 impact categories
11	BMW	2,165,566	LCA	ISO 14040/44	environmental	"Umwelterklärung"/ "Environmental Declaration" for selected models	- relative results showing distribution of CO_2e emissions over life cycle - life span: 150,000–250,000 km - focus on GHG emissions
12	Fiat	1,904,618	LCA	-	environmental	-	-
13	Daimler AG	1,808,125	LCA	ISO 14040/44	environmental	"Umweltzertifikat"/ "Environmental Certificate"	- absolute results - life span: 160,000–300,000 km - focus on CML2001 impact categories
14	SAIC	1,769,837	-	-	-	-	-
15	Changan	1,089,179			*[Only Chinese communication]*		
Subtotal Top 15		***61,014,905***					
World Total		**72,068,994**					

3.3. Implications for Application at an Automotive Company

Looking at the results from both research steps in this paper, it can be surmised that the structured literature analysis provided a view of the state of the art in the field of LCSA highlighting the major areas for research in the identified thematic clusters. Taking those findings and adding the insights gained in the second step of this research on the current status of product sustainability assessment in the automotive industry, the implications for the next steps towards the operationalization of LCSA were derived by consulting with decision makers within an automotive company. In the following, the possible next steps to address these research needs at an automotive company are proposed.

Several challenges for operationalization were identified that hinder the adoption of LCSA at an automotive company. A considerable barrier is the challenge to consistently carry out the three life cycle techniques. Within the thematic field of framework development there were still some unanswered challenges regarding the different maturities of the methods as well as questions regarding the requirement of consistent system boundaries. The systematical prerequisite of applying the same system boundaries is questioned because of the different foci that the individual methods have which could be understood as having no consistent system boundaries. This poses the question how consistent the system boundaries for practical implementation have to be. It can be argued that even the formally consistent formulation of system boundaries will yield emphases on different life cycle phases depending on the method carried out. For instance, even if research and development is an important part of the product life cycle from an economic perspective, it usually has marginal impacts on environment and society [15]. Heijungs et al. [88] understand this as the natural result of three different perspectives on the same object of interest. Decision makers saw the consistency of system barriers as less critical as long as it is ensured that the main impacts in every dimension were identified.

Proposed next step: For operationalization at an automotive company, a case study would be desirable that executes the LCSA with formally consistent boundaries and formats the seemingly inconsistent results in such a way that decision makers can nevertheless arrive at a conclusion.

As already mentioned, the low methodical maturity of S-LCA, notwithstanding the quasi-guidelines by UNEP/SETAC, and the more technical barrier of limited data availability for social impacts pose an additional barrier to implementing S-LCA as part of an LCSA framework at an automotive company as experiences regarding the social assessment of products at the 15 examined OEMs are rather scarce. Decision makers saw an entire S-LCA as too big a challenge for the time being but saw definitive advantage in social risk assessment of supply chains in order to efficiently manage social risks.

Proposed next step: When working towards operationalization of S-LCA, it should be assured that the indicators are relevant for companies operating in the automotive industry. Additionally, the relevancy for the coverage of social topics should match the strategic approach of the company as e.g., defined in their materiality analyses. This would enable linking corporate strategy and product sustainability even more closely. Looking at data availability, there currently are several efforts to build up social databases to address this issue, resorting more often than not to MRIO databases as basis, for instance the Social Hotspot Database or the Product Social Impact Life Cycle Assessment (PSILCA), which enable the simulation of supply chains and thus the building of a life cycle inventory even when limited data are available. A next step towards integration would be the evaluation of these databases and the decision for a mode in which data should be acquired, e.g., hotspot identification via databases and then primary data collection for said hotspots. Important insights might additionally be gained once the dissertation about S-LCA at Daimler is published.

Another methodical challenge not as prominently identified by the structured literature review but strongly emphasized by decision makers was the interpretation of results. As the support for decision makers is the intended goal of an LCSA study, clear and easily understandable results are desired. Therefore, aggregation and/or weighting of results for the different sustainability dimensions are topics that have received some attention with different results calling for further research.

Proposed next step: For adoption at an automotive company, this will be of crucial importance along with communication formats as the eventual goal of applying LCSA is to enable informed decisions to improve product sustainability. Additionally, the presentation and communication of results is an aspect that harbors the potential of becoming a barrier when done poorly but also being an enabler when used to its full potential. Therefore, it should be investigated whether decision makers need the aggregation of results into one dimension or if other modes of interpretation and communication would also be viable.

Another point strongly emphasized by decision makers was the desire to be able to express sustainability impacts in monetary terms, i.e., costs or revenue for the company, as this would facilitate the interpretation and inclusion of sustainability criteria in management decisions.

Proposed next step: A monetization method for translating product sustainability impacts into monetary units relevant to the company (costs or revenues) should be developed.

Opposite the identified challenges, there was seen potential for the translation of LCSA into practice at an automotive company.

First, the widespread acceptance and proliferation of LCA across the majority of top car manufacturers, as discovered in the second step of this research, shows that life cycle thinking and assessment of sustainability impacts is not foreign to the industry. Thus, the introduction of a life cycle based assessment framework that adds two more dimensions to the existing environmental dimension should, on the account of "understanding of the method", not be a hindrance. Decision makers understood and generally supported the combined assessment and management of environmental and social aspects of a product's life cycle.

Additionally, the automotive industry keeps track of components that are built into their cars and the materials they are composed of with the help of the International Material Data System (IMDS). Based on this system with the additional information on in-house production, all materials that are incorporated in a car can be accounted for and thus data gathering for the building of the physical life cycle inventory is facilitated.

Proposed next steps: At several OEMs, inter alia BMW, Daimler, and VW, information from the IMDS is already used for doing LCA. For prospectively using it to also conduct S-LCA, the expansion of information gathered with this system should be expanded to at least also contain information about the origins of materials and components.

3.4. Limitations

First and foremost, the search for literature on only four distinct sources and the stringent definition of the search term in order to narrow down the amount of initial hits pose as sources for uncertainty in the final set of relevant publications as important contributions could have been overlooked due to a too narrow search focus. Likewise, the exclusion of publications that addressed only one sustainability dimension can have an impact on the picture that this research yielded for the state of the art of LCSA as works that address advances of singular methods that are nevertheless part of LCSA did not find their way into the final assessment.

4. Conclusions

The aim of this paper was to identify challenges and potential for implementing Life Cycle Sustainability Assessment (LCSA) at an automotive company. The identified research needs regarding LCSA and the resulting next steps towards operationalization of LCSA at an automotive company were:

Consistent execution of the three life cycle based assessment methods

- Conduct case study to determine the best handling of the question around consistent system boundaries, especially whether a consistent picture of the results is needed.

Comparatively low maturity of S-LCA

- Ensure consistency of social topics with company specific strategy.
- Evaluate currently available databases and decide on a mode of data acquisition.

Presentation and interpretation of results

- Regarding aggregation of sustainability dimensions, it should be determined what weights decision makers would assign to the individual sustainability criteria or dimensions.
- A way of monetizing sustainability impacts should be developed to support decision makers in interpreting the results.
- A communication format should be devised that easily and clearly conveys the LCSA results and is directed at the relevant decision makers.

Most of these steps would ideally be taken by carrying out a case study at an automotive company after methodical questions, e.g., indicators for S-LCA, data acquisition mode, and way of aggregation, have been addressed. The proliferation of LCA within the automotive industry and the IMDS as common data acquisition system can facilitate the operationalization.

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References and Notes

1. JRC. *Towards a Life-Cycle Based European Sustainability Footprint Framework*; Pelletier, N., Maas, R., Goralcyk, M., Wolf, M.-A., Eds.; Publications Office of the European Union: Luxembourg, 2012.
2. Singh, R.K.; Murty, H.R.; Gupta, S.K.; Dikshit, A.K. An overview of sustainability assessment methodologies. *Ecol. Indic.* **2009**, *15*, 281–299. [CrossRef]
3. UNEP/SETAC Life Cycle Initiative. *Towards a Life Cycle Sustainability Assessment-Making Informed Choices on Products*; Valdivia, S., Ugaya, C.M.L., Sonnemann, G., Hildenbrand, J., Eds.; UNEP/SETAC Life Cycle Initiative: Paris, France, 2011.
4. Klöpffer, W. Life-Cycle based methods for sustainable product development. *Int. J. Life Cycle Assess.* **2003**, *8*, 157–159. [CrossRef]
5. Klöpffer, W. Life cycle sustainability assessment of products. *Int. J. Life Cycle Assess.* **2008**, *13*, 89–95. [CrossRef]
6. Finkbeiner, M.; Reimann, K.; Ackermann, R. Life cycle sustainability assessment (LCSA) for products and processes. In Proceedings of the SETAC Europe 18th Annual Meeting, Warsaw, Poland, 25–29 May 2008.
7. Finkbeiner, M.; Schau, E.M.; Lehmann, A.; Traverso, M. Towards Life Cycle Sustainability Assessment. *Sustainability* **2010**, *2*, 3309–3322. [CrossRef]
8. Ness, B.; Urbel-Piirsalu, E.; Anderberg, S.; Olsson, L. Categorising tools for sustainability assessment. *Ecol. Econ.* **2007**, *60*, 498–508. [CrossRef]
9. Zamagni, A. Life cycle sustainability assessment. *Int. J. Life Cycle Assess.* **2012**, *17*, 373–376. [CrossRef]
10. ISO 14040. *DIN EN ISO 14040: Umweltmanagement–Ökobilanz–Grundsätze und Rahmenbedingungen*; ISO: Geneva, Switzerland, 2006.
11. ISO 14044. *DIN EN ISO 14044: Umweltmanagement–Ökobilanz–Anforderungen und Anleitungen*; ISO: Geneva, Switzerland, 2006.
12. UNEP/SETAC. *Guidelines for Social Life Cycle Assessment of Products*; UNEP/Earthprint: Nairobi, Kenya, 2009.
13. Dreyer, L. Inclusion of Social Aspects in Life Cycle Assessment of Products. Ph.D.Thesis, Technical University of Denmark, Copenhagen, Danemark, 2009.

14. Bubeck, D. *Life Cycle Costing (LCC) im Automobilbau: Analyse und Planung von Lebenszykluskosten bei der Entwicklung von Produkten und Produktsystemen. Ansatz zur Integration des LCC Innerhalb der Ganzheitlichen Bilanzierung*, 1st ed.; Quantitative Methoden in Forschung und Praxis; Kovač: Hamburg, Germany, 2002.
15. Swarr, T.E.; Hunkeler, D.; Klöpffer, W.; Hanna-Leena, P.; Ciroth, A.; Brent, A.C.; Pagan, R. Environmental Life Cycle Costing: A Code of Practice. *Int. J. Life Cycle Assess.* **2011**, *16*, 389–391. [CrossRef]
16. WCED. *Report of the World Commission on Enviroment and Development: Our Common Future*; UN: New York, NY, USA, 1987.
17. Sala, S.; Farioli, F.; Zamagni, A. Life cycle sustainability assessment in the context of sustainability science progress (part 2). *Int. J. Life Cycle Assess.* **2013**, *18*, 1686–1697. [CrossRef]
18. Tranfield, D.; Denyer, D.; Smart, P. Towards a Methodology for Developing Evidence-Informed Management Knowledge by Means of Systematic Review. *Br. J. Manag.* **2003**, *14*, 207–222. [CrossRef]
19. Pickering, C.; Byrne, J. The benefits of publishing systematic quantitative literature reviews for PhD candidates and other early-career researchers. *High. Educ. Res. Dev.* **2014**, *33*, 534–548. [CrossRef]
20. Jasinski, D.; Meredith, J.; Kirwan, K. A comprehensive review of full cost accounting methods and their applicability to the automotive industry. *J. Clean. Prod.* **2015**, *108*, 1123–1139. [CrossRef]
21. Duff, A. The literature search: A library-based model for information skills instruction. *Libr. Rev.* **1996**, *45*, 14–18. [CrossRef]
22. Randolph, J. A Guide to Writing the Dissertation Literature Review. *Pract. Assess. Res. Eval.* **2009**, *14*, 1–13.
23. Petticrew, M.; Roberts, H. *Systematic Reviews in the Social Sciences—A Practical Guide*; Blackwell Publishing Ltd.: Oxford, UK, 2006.
24. Mattioda, R.A.; Mazzi, A.; Canciglieri, O.; Scipioni, A. Determining the principal references of the social life cycle assessment of products. *Int. J. Life Cycle Assess.* **2015**, *20*, 1155–1165. [CrossRef]
25. Zimmer, K.; Fröhling, M.; Schultmann, F. Sustainable supplier management—A review of models supporting sustainable supplier selection, monitoring and development. *Int. J. Prod. Res.* **2016**, *54*, 1412–1442. [CrossRef]
26. Miemczyk, J.; Johnsen, T.E.; Macquet, M. Sustainable purchasing and supply management: A structured literature review of definitions and measures at the dyad, chain and network levels. *Supply Chain Manag. Int. J.* **2012**, *17*, 478–496. [CrossRef]
27. Wacker, J. A definition of theory: Research guidelines for different theory-building research methods in operations management. *J. Oper. Manag.* **1998**, *16*, 361–385. [CrossRef]
28. Traverso, M.; Kim, P.; Brattig, S.; Wagner, V. *Managing Life Cycle Sustainability Aspects in the Automotive Industry*; Springer: Berlin, Germany, 2015; pp. 331–339.
29. Igarashi, M.; de Boer, L.; Fet, A.M. What is required for greener supplier selection? A literature review and conceptual model development. *J. Purch. Supply Manag.* **2013**, *19*, 247–263. [CrossRef]
30. Edmondson, A.C.; McManus, S.E. Methodological Fit in Management Field Research. *Acad. Manag. Rev.* **2007**, *32*, 1155–1179. [CrossRef]
31. Traverso, M.; Asdrubali, F.; Francia, A.; Finkbeiner, M. Towards life cycle sustainability assessment: An implementation to photovoltaic modules. *Int. J. Life Cycle Assess.* **2012**, *17*, 1068–1079. [CrossRef]
32. Traverso, M.; Finkbeiner, M.; Jørgensen, A.; Schneider, L. Life Cycle Sustainability Dashboard. *J. Ind. Ecol.* **2012**, *16*, 680–688. [CrossRef]
33. Sala, S.; Farioli, F.; Zamagni, A. Progress in sustainability science: Lessons learnt from current methodologies for sustainability assessment: Part 1. *Int. J. Life Cycle Assess.* **2013**, *18*, 1653–1672. [CrossRef]
34. Hoogmartens, R.; Van Passel, S.; Van Acker, K.; Dubois, M. Bridging the gap between LCA, LCC and CBA as sustainability assessment tools. *Environ. Impact Assess. Rev.* **2014**, *48*, 27–33. [CrossRef]
35. Klöpffer, W.; Grahl, B. From LCA to Sustainability Assessment. In *Life Cycle Assessment (LCA)*; Wiley-VCH Verlag GmbH & Co. KGaA: Weinheim, Germany, 2014; pp. 357–374.
36. Jørgensen, A.; Herrmann, I.T.; Bjørn, A. Analysis of the link between a definition of sustainability and the life cycle methodologies. *Int. J. Life Cycle Assess.* **2013**, *18*, 1440–1449. [CrossRef]
37. Keller, H.; Rettenmaier, N.; Reinhardt, G.A. Integrated life cycle sustainability assessment—A practical approach applied to biorefineries. *Appl. Energy* **2015**, *154*, 1072–1081. [CrossRef]
38. Neugebauer, S.; Martinez-Blanco, J.; Scheumann, R.; Finkbeiner, M. Enhancing the practical implementation of life cycle sustainability assessment-proposal of a Tiered approach. *J. Clean. Prod.* **2015**, *102*, 165–176. [CrossRef]

39. Cinelli, M.; Coles, S.R.; Jørgensen, A.; Zamagni, A.; Fernando, C.; Kirwan, K. Workshop on life cycle sustainability assessment: The state of the art and research needs—November 26, 2012, Copenhagen, Denmark. *Int. J. Life Cycle Assess.* **2013**, *18*, 1421–1424. [CrossRef]
40. Valdivia, S.; Ugaya, C.M.L.; Hildenbrand, J.; Traverso, M.; Mazijn, B.; Sonnemann, G. A UNEP/SETAC approach towards a life cycle sustainability assessment—our contribution to Rio + 20. *Int. J. Life Cycle Assess.* **2013**, *18*, 1673–1685. [CrossRef]
41. Hu, M.; Kleijn, R.; Bozhilova-Kisheva, K.P.; Di Maio, F. An approach to LCSA: The case of concrete recycling. *Int. J. Life Cycle Assess.* **2013**, *18*, 1793–1803. [CrossRef]
42. Stefanova, M.; Tripepi, C.; Zamagni, A.; Masoni, P. Goal and Scope in Life Cycle Sustainability Analysis: The Case of Hydrogen Production from Biomass. *Sustainability* **2014**, *6*, 5463–5475. [CrossRef]
43. Pesonen, H.-L.; Horn, S. Evaluating the Sustainability SWOT as a streamlined tool for life cycle sustainability assessment. *Int. J. Life Cycle Assess.* **2013**, *18*, 1780–1792. [CrossRef]
44. Foolmaun, R.K.; Ramjeawon, T. Life cycle sustainability assessments (LCSA) of four disposal scenarios for used polyethylene terephthalate (PET) bottles in Mauritius. *Environ. Dev. Sustain.* **2013**, *15*, 783–806. [CrossRef]
45. Foolmaun, R.K.; Ramjeeawon, T. Comparative life cycle assessment and social life cycle assessment of used polyethylene terephthalate (PET) bottles in Mauritius. *Int. J. Life Cycle Assess.* **2013**, *18*, 155–171. [CrossRef]
46. Vinyes, E.; Oliver-Solà, J.; Ugaya, C.; Rieradevall, J.; Gasol, C.M. Application of LCSA to used cooking oil waste management. *Int. J. Life Cycle Assess.* **2013**, *18*, 445–455. [CrossRef]
47. Souza, R.G.; Rosenhead, J.; Salhofer, S.P.; Valle, R.A.B.; Lins, M.P.E. Definition of sustainability impact categories based on stakeholder perspectives. *J. Clean. Prod.* **2015**, *105*, 41–51. [CrossRef]
48. Manzardo, A.; Ren, J.; Mazzi, A.; Scipioni, A. A grey-based group decision-making methodology for the selection of hydrogen technologies in life cycle sustainability perspective. *Int. J. Hydrog. Energy* **2012**, *37*, 17663–17670. [CrossRef]
49. Ghadimi, P.; Azadnia, A.H.; Mohd Yusof, N.; Mat Saman, M.Z. A weighted fuzzy approach for product sustainability assessment: A case study in automotive industry. *J. Clean. Prod.* **2012**, *33*, 10–21. [CrossRef]
50. Wang, X.; Chan, H.K. An integrated fuzzy approach for evaluating remanufacturing alternatives of a product design. *J. Remanuf.* **2013**, *3*, 10. [CrossRef]
51. Ren, J.; Manzardo, A.; Mazzi, A.; Zuliani, F.; Scipioni, A. Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multicriteria decision-making. *Int. J. Life Cycle Assess.* **2015**, *20*, 842–853. [CrossRef]
52. Buchert, T.; Neugebauer, S.; Schenker, S.; Lindow, K.; Stark, R. Multi-criteria Decision Making as a Tool for Sustainable Product Development–Benefits and Obstacles. *Procedia CIRP* **2015**, *26*, 70–75. [CrossRef]
53. Halog, A.; Manik, Y. Advancing Integrated Systems Modelling Framework for Life Cycle Sustainability Assessment. *Sustainability* **2011**, *3*, 469–499. [CrossRef]
54. Ostermeyer, Y.; Wallbaum, H.; Reuter, F. Multidimensional Pareto optimization as an approach for site-specific building refurbishment solutions applicable for life cycle sustainability assessment. *Int. J. Life Cycle Assess.* **2013**, *18*, 1762–1779. [CrossRef]
55. Bachmann, T.M. Towards life cycle sustainability assessment: Drawing on the NEEDS project's total cost and multi-criteria decision analysis ranking methods. *Int. J. Life Cycle Assess.* **2013**, *18*, 1698–1709. [CrossRef]
56. Kucukvar, M.; Tatari, O. Towards a triple bottom-line sustainability assessment of the U.S. construction industry. *Int. J. Life Cycle Assess.* **2013**, *18*, 958–972. [CrossRef]
57. Onat, N.; Kucukvar, M.; Tatari, O. Integrating triple bottom line input–output analysis into life cycle sustainability assessment framework: The case for US buildings. *Int. J. Life Cycle Assess.* **2014**, *19*, 1488–1505. [CrossRef]
58. Onat, N.; Kucukvar, M.; Tatari, O. Towards Life Cycle Sustainability Assessment of Alternative Passenger Vehicles. *Sustainability* **2014**, *6*, 9305–9342. [CrossRef]
59. Kucukvar, M.; Noori, M.; Egilmez, G.; Tatari, O. Stochastic decision modeling for sustainable pavement designs. *Int. J. Life Cycle Assess.* **2014**, *19*, 1185–1199. [CrossRef]
60. Schulz, M.; Short, M.D.; Peters, G.M. A streamlined sustainability assessment tool for improved decision making in the urban water industry. *Integr. Environ. Assess. Manag.* **2012**, *8*, 183–193. [CrossRef] [PubMed]
61. Nzila, C.; Dewulf, J.; Spanjers, H.; Tuigong, D.; Kiriamiti, H.; van Langenhove, H. Multi criteria sustainability assessment of biogas production in Kenya. *Appl. Energy* **2012**, *93*, 496–506. [CrossRef]

62. Luthe, T.; Kägi, T.; Reger, J. A Systems Approach to Sustainable Technical Product Design. *J. Ind. Ecol.* **2013**, *17*, 605–617. [CrossRef]
63. Mjörnell, K.; Boss, A.; Lindahl, M.; Molnar, S. A Tool to Evaluate Different Renovation Alternatives with Regard to Sustainability. *Sustainability* **2014**, *6*, 4227–4245. [CrossRef]
64. Stamford, L.; Azapagic, A. Life cycle sustainability assessment of electricity options for the UK. *Int. J. Energy Res.* **2012**, *36*, 1263–1290. [CrossRef]
65. Stamford, L.; Azapagic, A. Life cycle sustainability assessment of UK electricity scenarios to 2070. *Energy Sustain. Dev.* **2014**, *23*, 194–211. [CrossRef]
66. Dewulf, J.; Mancini, L.; Blengini, G.A.; Sala, S.; Latunussa, C.; Pennington, D. Toward an Overall Analytical Framework for the Integrated Sustainability Assessment of the Production and Supply of Raw Materials and Primary Energy Carriers. *J. Ind. Ecol.* **2015**, *19*, 963–977. [CrossRef]
67. Hirschberg, S.; Burgherr, P. Sustainability Assessment for Energy Technologies. In *Handbook of Clean Energy Systems*; John Wiley & Sons, Ltd.: Hoboken, NJ, USA, 2015; pp. 1–22.
68. Moriizumi, Y.; Matsui, N.; Hondo, H. Simplified life cycle sustainability assessment of mangrove management: A case of plantation on wastelands in Thailand. *J. Clean. Prod.* **2010**, *18*, 1629–1638. [CrossRef]
69. Zhou, Z.; Jiang, H.; Qin, L. Life cycle sustainability assessment of fuels. *Fuel* **2007**, *86*, 256–263. [CrossRef]
70. Ingwersen, W.; Cabezas, H.; Weisbrod, A.; Eason, T.; Demeke, B.; Ma, X.; Hawkins, T.R.; Lee, S.-J.; Bare, J.; Ceja, M. Integrated Metrics for Improving the Life Cycle Approach to Assessing Product System Sustainability. *Sustainability* **2014**, *6*, 1386–1413. [CrossRef]
71. Li, H.; Nitivattananon, V.; Li, P. Developing a Sustainability Assessment Model to Analyze China's Municipal Solid Waste Management Enhancement Strategy. *Sustainability* **2015**, *7*, 1116–1141. [CrossRef]
72. Azapagic, A.; Stichnothe, H. Life cycle sustainability assessment of biofuels. In *Handbook of Biofuels Production*; Luque, R., Ed.; Woodhead Publishing Ltd.: Cambridge, UK, 2010; pp. 37–60.
73. Jeswani, H.K.; Azapagic, A. Life cycle sustainability assessment of second generation biodiesel. In *Advances in Biodiesel Production*; Woodhead Publishing Ltd.: Cambridge, UK, 2012; pp. 13–31.
74. Schau, E.M.; Traverso, M.; Finkbeiner, M. Life cycle approach to sustainability assessment: A case study of remanufactured alternators. *J. Remanuf.* **2012**, *2*, 5. [CrossRef]
75. Lu, B.; Li, B.; Wang, L.; Yang, J.; Liu, J.; Wang, X.V. Reusability based on Life Cycle Sustainability Assessment: Case Study on WEEE. *Procedia CIRP* **2014**, *15*, 473–478. [CrossRef]
76. Yu, M.; Halog, A. Solar Photovoltaic Development in Australia—A Life Cycle Sustainability Assessment Study. *Sustainability* **2015**, *7*, 1213–1247. [CrossRef]
77. Martínez-Blanco, J.; Lehmann, A.; Muñoz, P.; Antón, A.; Traverso, M.; Rieradevall, J.; Finkbeiner, M. Application challenges for the social Life Cycle Assessment of fertilizers within life cycle sustainability assessment. *J. Clean. Prod.* **2014**, *69*, 34–48. [CrossRef]
78. Minne, E.; Crittenden, J.C. Impact of maintenance on life cycle impact and cost assessment for residential flooring options. *Int. J. Life Cycle Assess.* **2015**, *20*, 36–45. [CrossRef]
79. Brandão, M.; Clift, R.; Canals, L.M.; Basson, L. A Life-Cycle Approach to Characterising Environmental and Economic Impacts of Multifunctional Land-Use Systems: An Integrated Assessment in the UK. *Sustainability* **2010**, *2*, 3747–3776. [CrossRef]
80. Saling, P.; Kicherer, A.; Dittrich-Krämer, B.; Wittlinger, R.; Zombik, W.; Schmidt, I.; Schrott, W.; Schmidt, S. Eco-efficiency analysis by basf: The method. *Int. J. Life Cycle Assess.* **2002**, *7*, 203–218. [CrossRef]
81. Schmidt, I.; Meurer, M.; Saling, P.; Kicherer, A.; Reuter, W.; Gensch, C.-O. SEEbalance. *Green. Manag. Int.* **2004**, *2004*, 78–94. [CrossRef]
82. Saling, P.; Maisch, R.; Silvani, M.; König, N. Assessing the Environmental-Hazard Potential for Life Cycle Assessment, Eco-Efficiency and SEEbalance (8 pp). *Int. J. Life Cycle Assess.* **2005**, *10*, 364–371. [CrossRef]
83. Grießhammer, R.; Buchert, M.; Gensch, C.; Hochfeld, C.; Manhart, A.; Rüdenauer, I.; Ebinger, F. *PROSA—Product Sustainability Assessment*; Öko-Institut e.V.: Berlin, Germany, 2007.
84. Fontes, J.; Bolhuis, A.; Bogaers, K.; Saling, P.; van Gelder, R.; Traverso, M.; Das Gupta, J.; Morris, D.; Bosch, H.; Woodyard, D.; et al. *Handbook for Product Social Impact Assessment*, 2nd ed.; PRé Sustainability: Amersfoot, Holland, 2015.
85. OICA. Production Statistics 2014. Available online: http://www.oica.net/category/production-statistics/2014-statistics/ (accessed on 15.08.2015).

86. Karlewski, H. Social Life Cycle Assessment in the Automotive Industry. Doctoral Thesis, Technische Universität Berlin, Berlin, German, 2016.
87. European Commission Directive 2000/53/EC on end-of life vehicles. *Off. J. Eur. Communities* **2000**, *L 269*, 1–15.
88. Heijungs, R.; Huppes, G.; Guinée, J.B. *A Scientific Framework for LCA*; Institute of Enviromental Science, Leiden Unversity: Leiden, The Netherlands, 2009.

1.5 Goal of the Thesis and Research Objectives

The goal of this dissertation is to increase the applicability of LCSA at an automotive company to enable the identification and implementation of product sustainability improvement potential. That means, the focus lies on filling some of the gaps of the existing LCSA methodology rather than developing new methodological approaches. The main impediments that need to be addressed to put LCSA into practice at an automotive company were identified in section 1.4. To overcome these impediments, the research objectives of this thesis were defined to address the corresponding research needs of section 1.4. The following research objectives were set for this thesis:

- **Research Objective 1**: Approach to select social indicators and to focus primary data collection in S-LCA
    - Selection of social topics relevant to the company and deduction of corresponding indicators
    - Evaluation of currently available databases for social hotspot assessment
    - Examination of additional options for social hotspot assessment to streamline primary data collection
- **Research Objective 2**: Method to interpret LCSA results for decision makers
    - Determine what weights decision makers would assign to the individual sustainability dimensions
    - Choose a weighting set to adopt in the LCSA framework
- **Research Objective 3**: Mechanism to implement LCSA results in decision process
    - Development of a method to monetize product sustainability impacts
    - Devise a mechanism to integrate findings in the product-related decision process at an automotive company

The structure of this thesis is presented in the next section.

1.6 Thesis Structure

The introduction and background along with the goal of this thesis have been presented in section 1. They are followed by the presentation of the results, most of which were published beforehand in separate papers (section 2). In section 3, the results are discussed. The conclusions of this dissertation together with an outlook are given in section 4. Figure 4 shows the structure of this thesis including the integration of the previously published results.

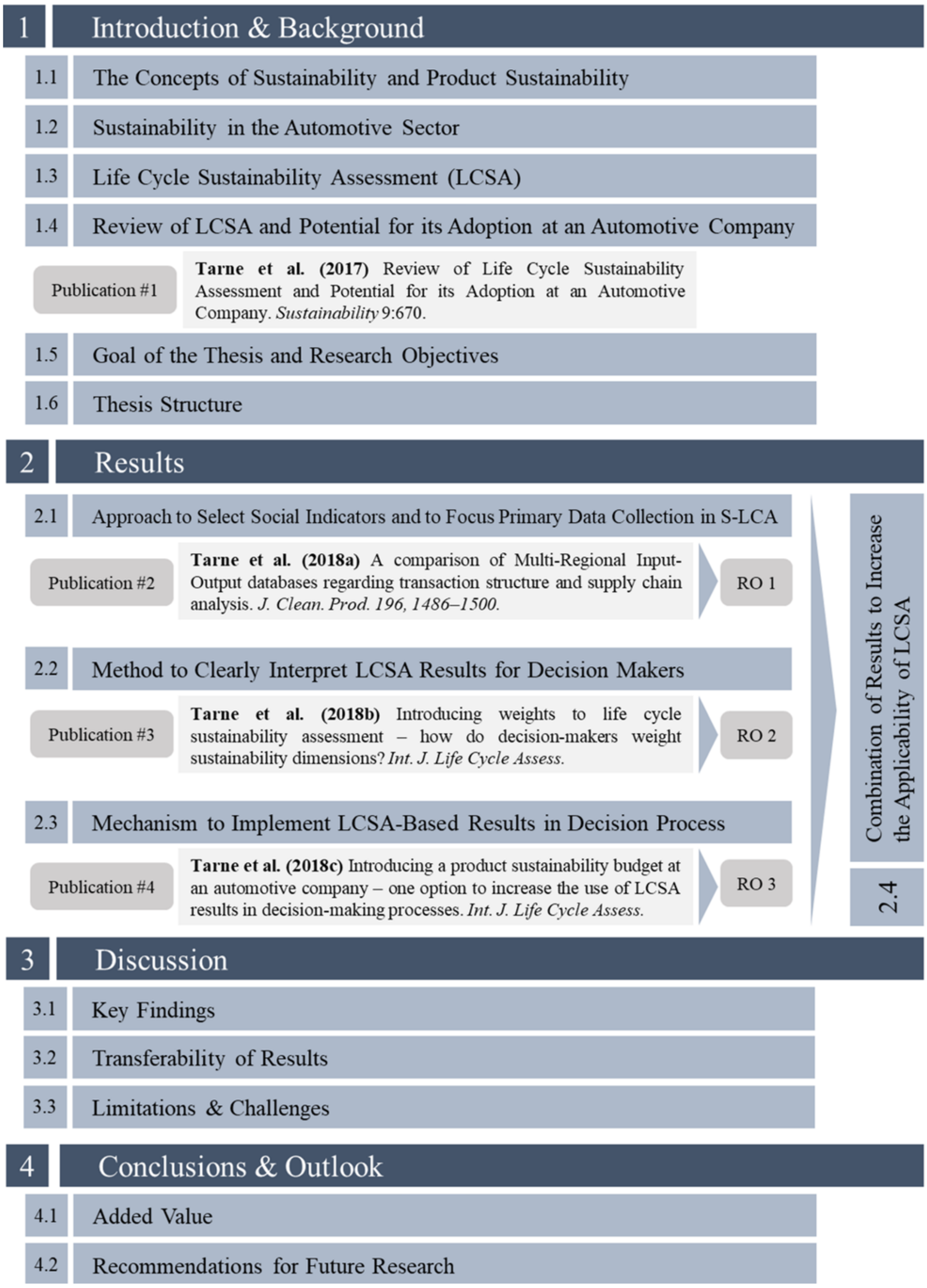

Figure 4: Thesis structure

It can be seen from Figure 4 that the first publication — a review paper — contributes to the establishing the state of the art and the identification of research gaps regarding LCSA. The other three publications address the specific research objectives (RO). The results of this thesis are presented in the next section.

2 Results

This section presents the results of the thesis and shows how the individual publications combined in this research tie into the overarching goal: Increasing the applicability of LCSA at an automotive company to enable the identification and implementation of product sustainability improvement potential. In section 2.1, the results regarding the selection of social topics, indicators and the focusing of primary data collection for S-LCA (research objective 1) are presented. The findings on an interpretation of LCSA results for decision makers (research objective 2) are presented in section 2.2. Section 2.3 contains the findings on the implementation of LCSA results in product related decision processes at a company (research objective 3). In section 2.4 the results from sections 2.1 – 2.3 are combined and it is shown how their combination helped in making the LCSA framework more applicable for the automotive industry.

As the intention was to increase the applicability of LCSA at an automotive company, the LCSA framework was applied to the component level of a vehicle. This granularity of assessment was chosen because components are often used as the smallest manageable unit at automotive companies and there is an increasing trend of outsourcing component development to suppliers (Ciravegna et al., 2013). This application to the component level has first been introduced by Tarne et al. (2018) and is briefly explained in the following as it formed the bearing for combining the individual research papers into one dissertation.

Table 1: Challenges for applying the LCSA framework to the component level of a vehicle

Vehicle Component	LCA	LCC	S-LCA	Overall LCSA
A	?	?	?	?
B	?	?	?	?
C	?	?	?	?
...	?	?	?	?

Table 1 shows the challenges that had to be overcome for an application of the LCSA framework to the component level of a vehicle. Every component of the entire vehicle would be listed and evaluated based on its impact on every LCSA dimension measured with the underlying methods LCA, LCC and S-LCA. That means, the impact of the production, use and end of life of the respective component on all three sustainability dimensions is evaluated and combined into an overall LCSA impact.

With the help of this application of the LCSA framework to the component level, an overall LCSA evaluation per component should be possible and this approach should enable practitioners to:

a) Identify the component with the highest overall impact on product sustainability

b) Evaluate, whether improvement measures for individual components or the entire car are preferable when looking at the overall LCSA impact

When these two functionalities of the LCSA framework are ensured, the goal of identifying and implementing product sustainability improvements at an automotive company can be achieved. To increase the applicability of this LCSA framework, this thesis tries to remove the question marks in table 1 by addressing the research objectives as defined in section 1.5. To achieve this, the following steps were undertaken:

1. Assessment of the LCA and LCC impacts of components
2. Expression of the impacts as criticality points
3. Selection of indicators and data basis for S-LCA
4. Assessment of the S-LCA impact of components and expression as S-LCA criticality points
5. Combination of criticality points from LCA, LCC and S-LCA into one LCSA criticality score

In the following, these steps are shown in detail. The impact in the individual dimensions could be determined based on any selection of indicators deemed relevant. For instance, a car manufacturer might have identified addressing climate change in its sustainability strategy. Therefore, the GWP would be an indicator to select for evaluating the environmental impact of its products. The separate indicator results could then combined to an overall (negative) impact in each dimension. As different indicators will have different units of measurements, e.g. kg of CO_2e emitted or m^3 of water consumed, the overall result in each dimension should be dimensionless. For the appraisal of the economic impact of singular components in the use phase, the frequency of replacement would have to be ascertained and how many of those fall under warranty, meaning the manufacturer bears the cost, and how many would fall into the regular aftersales market, meaning the customer bears the cost. The overall impact of every component should contain the impacts due to the production, use and end of life (EoL) of that component. The impact of the production of every component can clearly be determined. The impact of assembly of the car can be included as separate entry in the component list or be allocated to the components. This could be done via allocation by mass (or the respective process like welding for car frame, painting to car body etc.). The impacts of the use phase related to maintenance and exchanging wear and tear parts can again be clearly determined per component. The impacts of the use phase related to the provision and use of the fuel can also be allocated to the components, e.g. via mass or even depending on the impact that the component has on the drag factor. Impacts on EoL can again be clearly determined as parts for remanufacturing and recycling get dissembled and the rest of the car gets shredded. This research applied a mass allocation approach, meaning that the impacts of the vehicle's use phase were allocated to the individual components relative to their share in the entire vehicle's weight. In the first step, the impacts of every component were determined by LCA and LCC. This step was not derived in the individual research papers but was developed by applying LCSA at an automotive company. As the environmental and economic dimensions were already implemented at the concerned company, the established indicators, i.e. greenhouse gases, energy consumption, water consumption and cost, were used to assess the LCA and LCC impacts within the LCSA framework. In the second step, the LCA and LCC impacts were expressed as the criticality points. Criticality points are dimensionless and have been created in the course of this approach, i.e. they are not based on any previous or external sources. They express the relative criticality of a component in the respective dimension, meaning that a component with a high count of criticality points has a more negative impact than a component with a low count of criticality points. For this kind of relative evaluation, outranking methods like VIKOR (Opricovic & Tzeng, 2004), PROMETHEE (Brans, 1982; Behzadian et al., 2010) or ELECTRE (Benayoun et al., 1966; Govindan & Jepsen, 2016) would be suitable as they can combine indicators with different measurement units. For this relative evaluation, the VIKOR method (Opricovic & Tzeng, 2004) was chosen. To determine the relative impacts of a component on the LCA dimension based on the three indicator results for GWP, energy consumption and water consumption, the following steps were taken in accordance with the VIKOR method (Opricovic & Tzeng, 2004, 2007). First, the S_j and R_j values were calculated representing the values for the strategy of maximum group utility (S) and individual regret (R), respectively.

$$S_j = \sum_{i=1}^{n} w_i (f_i^* - f_{ij})/(f_i^* - f_i^-) \tag{2}$$

S_j = S-value for component j

w_i = weight of indicator i

f_{ij} = result of indicator i for component j

f_i^*= highest result of indicator i over all components

f_i^- = lowest result of indicator i over all components

n = number of components

The weight of indicator i (w_i) was set as one for all three indicators leading to an equal weighting between all three.

$$R_j = max_i \left[\sum_{i=1}^{n} w_i (f_i^* - f_{ij})/(f_i^* - f_i^-) \right] \tag{3}$$

R_j = R-value for component j

Afterwards, the Q_j value per component was determined, balancing the group utility against the individual regret.

$$Q_j = v \frac{S_j - S^*}{S^- - S^*} + (1 - v) \frac{R_j - R^*}{R^- - R^*} \tag{4}$$

Q_j = Q-value for component j

S^* = $\min_j$ [S_j]

S^- = $\max_j$ [S_j]

R^* = $\min_j$ [R_j]

R^- = $\max_j$ [R_j]

v = weight for the strategy of maximum group utility

$1 - v$ = weight of the individual regret

The value for v was set at 0.75 to put the emphasis on the maximum group utility. The component with the highest overall impact in a specific LCSA dimension of the entire vehicle was awarded 100 criticality points, whereas the component with the lowest impact was assigned zero criticality points. Therefore, the Q_j value was multiplied by 100 to calculate the criticality points for every component j. Table 2 shows what the conceptual approach to increasing the applicability of LCSA looks like after the first two steps have been taken.

Table 2: LCSA framework with LCA and LCC impacts expressed as criticality points

Component	**LCA Criticality Points**	**LCC Criticality Points**	**S-LCA Criticality Points**	**Overall LCSA Criticality Points**
A	72	11	?	?
B	82	17	?	?
C	20	52	?	?
...	?	?	?	?

In this section, the LCSA framework has been applied to the component level of a vehicle and the concept of criticality points has been established. The next step in increasing the applicability of the LCSA framework is the selection of social indicators and the focusing of the primary data collection for S-LCA. This step is detailed in section 2.1.

2.1 Approach to Select Social Indicators and to Focus Primary Data Collection in S-LCA

The first research objective was addressed through the definition of social indicators and the attempt of improving social hotspot assessment, i.e. the identification of supply chain sections with increased risk of violations pertaining to relevant social topics. When social hotspots are known, the time and resource consuming task of primary data collection can be focused on the most critical parts of the supply chain, thus making S-LCA more practical to conduct. This section addressed the research objective 1 (c.f. section 1.5) as follows:

- **Selection of social topics relevant to the company and deduction of corresponding indicators**
  Section 2.1.1 contains the selection of social topics that are in line with the company specific strategy and the derivation of respective indicators.
- **Evaluation of currently available databases for social hotspot assessment**
  Multi-Regional Input-Output (MRIO) databases were examined regarding their reliability and robustness to apply them for social hotspot assessment. This was done because it was of interest to investigate, whether MRIO analysis could make S-LCA easier by enhancing social hotspot assessment through gaining supply chain transparency despite limited knowledge of suppliers (section 2.1.2).
- **Examination of additional options for social hotspot assessment to streamline primary data collection**
  It was found that MRIO databases did not offer robust results for conducting social hotspot assessment rendering them not suitable to guide data acquisition. Therefore, an alternative approach was developed, which identified social hotspots based on material supply chains and their social risks (section 2.1.3).

The selection of social topics and indicators relevant to the company was carried out to lay the basis for the assessment of the S-LCA dimension in the LCSA framework. This step was independent of the other to. The evaluation of MRIO databases for social hotspot assessment was carried out to investigate, whether they could be used to facilitate the initial steps in an S-LCA. Because they did not offer the support for social hotspot assessment as it was hoped for, an additional third step was carried out. This third step examined an additional option to conduct social hotspot assessment that was based on material supply chains.

2.1.1 Selection of Social Topics and Indicators Relevant to the Company

To derive meaningful indicators for the assessment of social impacts at an automotive company, the identification and prioritization of material topics[12] was carried out. In the course of sustainability reporting, companies compile a materiality matrix in which they rank the relevance of sustainability topics from their point of view against the importance of these topics for external stakeholders (Whitehead, 2017). Figure 5 shows the material social topics as defined in the BMW Sustainable Value Report (SVR). The BMW Group determines the materiality matrix by conducting stakeholder dialogue meetings and gathering the expectations of stakeholders' general priorities for sustainability for the company. For the 2014 materiality matrix, the process involves telephone interviews with representative stakeholders, including customers, suppliers, investors, authorities, NGOs and scientists from different regions of the world. The interview results were combined with the results of the stakeholder survey and analysis from 2012 (BMW Group, 2014). The internal perspective was gathered via a materiality workshop with 18 representatives from the relevant company divisions (including their strategy offices). In addition, a continuous dialogue with stakeholder groups is maintained (BMW Group, 2014). In the process, the material topics were classified regarding their relevance to the stakeholders and to the BMW Group. The relevance was expressed on a scale from zero (= no relevance) to nine (= high relevance). As threshold for inclusion of social topics, the relevance value of seven for BMW Group was chosen as the motivation of this research was to lay the focus on the most relevant social topics. Therefore, a value clearly above the level for medium relevance (4.5) was deemed appropriate. This led to the top four identified social topics (human rights, social standards in the supply chain, occupational health and safety, and corruption).

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[12] "Materiality is the threshold at which aspects become sufficiently important that they should be reported. Beyond this threshold, not all material aspects are of equal importance and the emphasis within a report should reflect the relative priority of these material aspects. In financial reporting, materiality is commonly thought of as a threshold for influencing the economic decisions of those using an organization's financial statements, investors in particular. The concept of a threshold is also important in sustainability reporting, but it is concerned with a wider range of impacts and stakeholders. Materiality for sustainability reporting is not limited only to those aspects that have a significant financial impact on the organization." (Global Reporting Initiative, 2015)

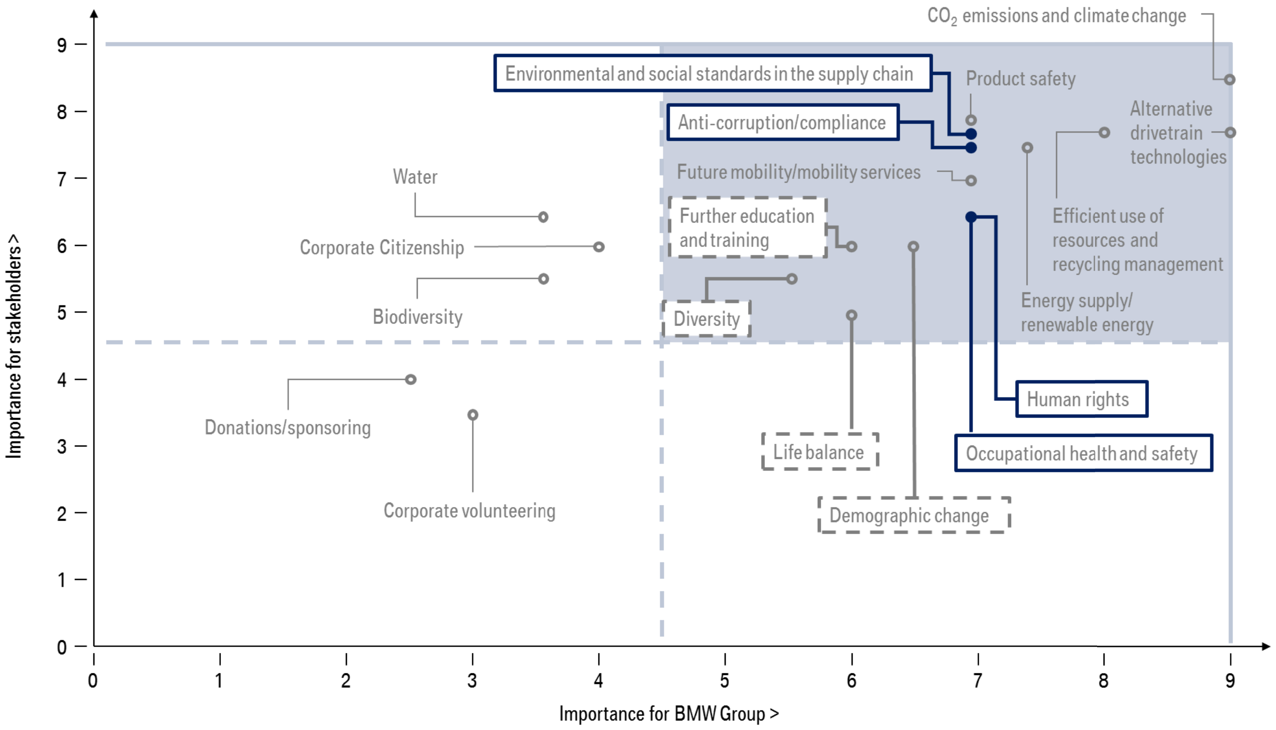

Figure 5: Social topics identified from the sustainability materiality matrix — own visualization after BMW Group (2014). The top right corner marks the area with the most material topics, i.e. topics with high importance for BMW Group as well as external stakeholders. Within this area, the topics that relate to the social dimension have been marked and categorized depending on whether they were identified as material topic for the S-LCA dimension within LCSA (dark blue) or not (grey).

For those social topics, indicators were defined. This was done by analyzing the indicator list contained in the SHDB Social Hotspot Database (SHDB) and selecting those that matched with the identified four social topics. The SHDB collects different international data from sources like ILO, Laborsta or the World Bank on social topics like child labor, forced labor or labor rights. The data is then interpreted and characterized to give a classification of the risk level (low, medium, high and very high) in the respective country and industry sector (Benoit-Norris et al., 2012; Benoît Norris, 2014). This assessment is made for all industry sectors as contained in the Global Trade Analysis Project (GTAP) for 191 countries (Benoit-Norris et al., 2012). For example, the statistical occurrence of child labor of 5% in a country and industry sector was characterized as medium risk (Benoít Norris et al., 2013) meaning that a process/company located in the respective country and industry sector can be regarded as "medium risky" to have instances of child labor if no additional information is at hand. The following list of indicators for the four topics resulted from the analysis of the SHDB:

1. **Human rights**
    - Risk of child labor
    - Risk of forced labor
    - Risk of violation of right to collective bargaining
    - Risk of violation of right of freedom of association
2. **Social standards in the supply chain**
    - Potential of Average wage being < Minimum Wage
    - Risk of excessive working time
3. **Occupational Health and Safety**
    - Risk of non-fatal injuries
    - Risk of fatal injuries
4. **Anti-Corruption / Compliance**
    - Overall Risk for Corruption

In this section, the social topics and corresponding indicators relevant to an automotive company to assess the impacts on the S-LCA dimension within the LCSA framework were determined. But the approach of how to actually go about the collection of data to actually determine the impacts on these social topics has still to be addressed. In the next section, MRIO databases are evaluated with respect to their possible use in social hotspot assessment. This would have the potential to streamline the efforts for primary data collection in S-LCA.

2.1.2 Comparison of MRIO databases regarding transaction structure and SC analysis

As MRIO databases contain the world's trade flows, they are often used to simulate supply chains for which no other information than the final demand or tier 1, i.e. the direct suppliers, exist (Kucukvar et al., 2014)(Onat et al., 2014). Therefore, MRIO databases were seen as an interesting option to enable social hotspot assessment, especially as they have already been used for conducting social risk assessment (Moran et al., 2015) or for constructing the SHDB (Benoit-Norris et al., 2012; Benoît Norris, 2014). Of interest was to find out how consistent the information of different MRIO databases are and how they should ideally be applied for SC analysis in order to receive useable results. That means, to obtain reliable information on the region and industry sector of all relevant suppliers in the respective SC for which only tier 1 is known. This would enable a social hotspot assessment to identify the most critical parts of an SC. This, in turn, could then streamline the efforts for primary data collection of S-LCA as the practitioner can focus on these hotspots for primary data collection. This would greatly reduce the effort for conducting S-LCA, significantly improving its applicability and thus the applicability of LCSA. The four major MRIO databases (Eora, EXIOBASE, GTAP and WIOD) were evaluated with regard to their comparability and consistency as well as to the best way to apply them to enable SC analysis. This was done by comparing their transaction structures, i.e. the amount of intermediate consumption that is transferred between regions and industry sector, and their SC analysis results, i.e. the information on industry sector and region of potential sub-suppliers of the supplier of interest. Transaction structures were compared based on a unified form to which all databases were transformed. Two types of SC analysis were carried out with all four databases: an aggregated SC analysis, i.e. based on the Leontief Inverse, and a disaggregated SC analysis, i.e. based on Structural Path Analysis. The results were also compared with a real SC. Results showed that this MRIO analysis did not offer robust results for social hotspot assessment as the supply chain modelling was not satisfying. Not satisfying means in this context that the SC modelled by the MRIO databases varied significantly between each other and also the real world SC that they were supposed to model. Therefore, MRIO analysis is not suitable to depict the real SC and can consequently not identify possible hotspots. The closest results to the real life case study were gained from the aggregated supply chain analysis. Following these results, MRIO were discarded as possible way of improving the applicability of S-LCA and the prioritization of data acquisition. Therefore, an alternative approach for identifying social hotspots has been developed within this dissertation, which is presented in section 2.1.3.

The publication by Tarne et al. (2018a) is presented on pages 48–63.

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In this section the possible application of MRIO analysis for supporting social hotspot assessment has been ruled out for the further course of the thesis. Therefore, another approach was examined that looked at the social risks related to the worldwide material production of materials relevant to the automotive industry. The approach is presented in the next section.

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A comparison of Multi-Regional Input-Output databases regarding transaction structure and supply chain analysis

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ABSTRACT

Multi-Regional Input-Output (MRIO) databases might be used to derive information of the regional and sectoral distribution of otherwise unknown supply chains (SC). There are several MRIO databases available that were created using different approaches. This paper evaluates MRIO databases with regard to (1) their comparability and consistency in general and (2) the best way to apply them to enable SC analysis. Four MRIO databases were analyzed: Eora, EXIOBASE, GTAP and WIOD. They were compared based on (A) transaction structure and (B) SC analysis results. (A) was done by transforming the databases to a unified form and determining the relative differences of their transaction values. For (B), a SC analysis based on cumulated intermediate consumption (CIC) and one based on structural path analysis (SPA) were carried out and compared with the respective real life SC. The results showed that around 80% of the total transaction volume in MRIO databases was intra-regional transfers. Inter-regional transfers varied strongly between databases but overall similarity between databases was high when considering the total transaction volume. This was supported by SC analysis based on CIC. The SC analysis by production tier varied significantly between databases with only 50–63% of simulated suppliers being from the same region or sector. The regional distribution of value added also showed clear differences between databases. The indication by the case study showed that the analysis based on CIC achieved more overall conformity with the real SC but both ways of analysis only capture a fraction (14%) correctly. These results led to the conclusion that the preferred mode of MRIO analysis for supporting subsequent SC assessments would be based on CIC but that neither mode would be recommended for detailed SC analysis as shown for the example of Social LCA. Future research should try to look at comparing disaggregated MRIO databases, analyze the creation processes of MRIO databases and clarify why the share of intra-regional transactions was consistent across databases but surprisingly high.

1. Introduction & goals

Companies are increasingly held accountable for the social and environmental impacts along their entire supply chain (SC) (Betts et al., 2018; Parmigiani et al., 2011; Wolf, 2014). Increasing transparency is a prerequisite for companies to assume responsibility along their SCs because to them mostly only tier 1 suppliers are known (Akkermans et al., 2004; Karlewski, 2016; Koplin et al., 2007; Lamming et al., 2001). For a large supply network of a complex product, a car for instance, this poses a practical challenge as the countless SCs easily encompass more than 10,000 suppliers – the BMW Group had around 13,000 in 2015 (BMW Group, 2015).

Especially comprehensive product sustainability assessment needs to address the impact of a product on all three dimensions of sustainability over its entire life cycle (Finkbeiner et al., 2010, 2008; Klöpffer, 2008, 2003; UNEP/SETAC Life Cycle Initiative, 2011). To be able to fully carry out such an analysis, the implementation of Social LCA (S-LCA) in product sustainability assessment at companies would be needed, which is still lagging, inter alia, due to still missing standardization of the method and limited data availability (Karlewski, 2016; Klöpffer and Grahl, 2014; UNEP/SETAC Life Cycle Initiative, 2011). The issue of data availability in S-LCA is challenging because social impacts can differ depending on the region and industry sector where the process takes place (Benoît Norris, 2014) as well as the company it takes place in (Dreyer et al.,

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2006; Jørgensen et al., 2010, 2008; Martínez-Blanco et al., 2015).

When trying to carry out hotspot assessments effectively for a specific SC, i.e. identifying the most critical parts of an SC in regards to possible violations the region and industry sector where the production steps are located have to be known. Then an appraisal of the risk for impact can be carried out, in the case of social violations, for instance, by resorting to social risk databases. These hotspots can give indication on where to focus primary data collection for more substantial risk assessments.

Multi-Regional Input-Output (MRIO) databases might deliver information on an otherwise unknown SC as they have been used to determine from which region in the world and which industry sector the required intermediate consumption of a defined output came from (Crama et al., 1984; Defourny and Thorbecke, 1984; Waugh, 1950). MRIO databases have been applied to assess the environmental burden of supply chains by combining their information on intermediate consumption with information on the environmental performance of industry sectors (Lenzen et al., 2013; Timmer et al., 2015; Wood et al., 2014) and are increasingly used to enhance the operability of life cycle based sustainability assessment of products (Kitzes et al., 2017; Kucukvar and Tatari, 2013; Moran et al., 2015; Onat et al., 2014a, 2014b). MRIO databases have also been used as information sources to enable product related social hotspot assessment of supply chains by creating socially extended MRIO databases (Benoit-Norris et al., 2012; Benoít Norris et al., 2013; Ciroth and Eisfeldt, 2015). The latest example is the approach by Zimmer et al. (2017) that built a social extension for Eora and combined it with supply chain management requirements at an automotive company. Even though MRIO databases are increasingly used in connection with LCA and several databases are available, so far comparative assessments of MRIO databases have mainly been carried out based on their environmental extensions (Bouwmeester and Oosterhaven, 2013; Lenzen, 2011; Lenzen et al., 2004; Moran and Wood, 2014; Owen et al., 2016, 2014; Stadler et al., 2014; Steen-Olsen et al., 2014; Su et al., 2010; Wyckoff and Roop, 1994). Regarding their transaction flows and their reliability and applicability for enhancing supply chain transparency, there is still need for further analysis (Arto et al., 2014; Steen-Olsen et al., 2016; Zimmer et al., 2016). Furthermore, the applicability of MRIO analysis to enhance supply chain transparency within an industry that produces a complex product has not yet been investigated and the reliability of MRIO analyses results should be ascertained. In the course of this paper “MRIO database“ refers to the actual databases while “MRIO analysis“ refers to the application of these databases, e.g. for SC analysis.

The motivation of this research is to find out how similar and consistent the information of different MRIO databases are and how they should ideally be applied for SC analysis in order to receive useable results, i.e. reliable information on the region and industry sector of all relevant suppliers in the respective SC. This research therefore aims at answering the following research questions:

1. How similar and consistent are the information within different MRIO databases?
2. How should MRIO analyses be applied in order to provide useful information on suppliers in a SC of interest?

The first question is addressed by comparing and analyzing the structures of MRIO databases. The second question is addressed by comparing and validating the results of different SC analyses. The approach of this paper is therefore also twofold: as a first step, four selected MRIO databases are compared based on the transaction flows (c.f. section 3.1). In a second step, two ways of analyzing a SC are tested between the four databases (c.f. section 3.2): the broad analysis of the overall distribution of suppliers based on the cumulated intermediate consumption (CIC) as determined by the Leontief Inverse (c.f. section 3.2.1) and the more detailed distribution of suppliers through the disaggregated demand by SC tiers, i.e.“unraveled” Leontief Inverse or structural path analysis (SPA) (c.f. section 3.2.2). The robustness of the results is additionally tested by exchanging the initial industry sector for the SC analysis of one database (i.e. the tier 1 industry sector is switched). Finally, a comparison of the two ways of SC analysis of the four different databases with a real life case study is conducted to get an indication on the transferability of SC (c.f. section 3.3). The usefulness of the results for an actual application is discussed using the exemplary application of the SC analysis results for a possible social hotspot assessment that could subsequently facilitate conducting an S-LCA (c.f. section 4.3.4).

2. Background

2.1. MRIO databases

MRIO databases contain information on the economic links between regions and industry sectors (Tukker and Dietzenbacher, 2013; Wiedmann et al., 2011). They are large tables that contain the monetary flows of goods and services that are purchased by one industry sector in a given region from another industry sector in the same or a different region (Lenzen et al., 2012). They are compiled by matching together supply and use tables (SUT) of the respective regions, i.e. specific balances of product and service supply and demand, and harmonizing them with the trade balances between regions to form a consistent picture of the world spanning transaction flows (Lenzen et al., 2013; Wiedmann et al., 2011). The first MRIO database goes back to Leontief and Strout (1963).

The **explicit form** of an MRIO database contains all absolute transaction volumes. In order to find the relative link between industry sector outputs and the intermediate consumption the **coefficient form** is derived. This form holds coefficients denoting the share of intermediate consumption required for providing a final product or service normalized to one currency unit.

There are several MRIO databases available, e.g. the Asian International Input-Output Table (AIIOT), Eora, EXIOBASE, GTAP, OpenEU, OECD and WIOD (Inomata and Owen, 2014; Peters et al., 2011). For this study Eora, EXIOBASE, GTAP and WIOD were selected, because they are the most frequently used databases. Some of their main features are summarized in Table 1.

2.1.1. Eora

The Eora database (Lenzen et al., 2013, 2012) was built with the aim of including as much information of the original data sources as possible. This was achieved by combining different information formats of different countries meaning that for one country the information was contained in supply and use tables (SUT) whereas for another country product-by-product tables were used. The database covers 189 regions and considers itself exhaustive in regards to the coverage of global Gross Domestic Product (GDP). An elaborate way of aggregating these information into 26 industry sectors was established as well as a back- and forecasting algorithm to deduce a time series spanning from 1990 to 2011. Additionally, Eora contains information on basic prices, margins and taxes.

2.1.2. EXIOBASE

The EXIOBASE development (Wood et al., 2014) focused on the European region and put its efforts into providing high resolution of environmentally sensitive industries like the energy sector. The database was built from SUT data from Eurostat for 27 EU countries and data for 16 additional countries from various sources. These

Table 1
Overview of the selected MRIO databases.

Name	No. of regions	No. of industry sectors	No. of transactions in the intermediate consumption matrix	Coverage of global GDP	Source
Eora	189	26	24,147,396	≈ 100%	(Lenzen et al., 2013)
EXIOBASE	48	163	61,214,976	>90%	(Wood et al., 2014)
GTAP	140	57	63,680,400	>98%	(Narayanan et al., 2012)
WIOD	41	35	2,059,225	>85%	(Timmer et al., 2015)

GDP = Gross Domestic Product.

Table 2
Aggregation of industry sectors to harmonize the dimension of the MRIO databases for comparability.[1]

No.	Unified industry sector name	Eora	EXIOBASE	GTAP	WIOD
1	Agriculture, hunting, forestry and fishing	1–2	1–19	1–14	1
2	Mining and Quarrying	3	20–34	15–18	2
3	Food & Beverages	4	35–48	19–26	3
4	Textiles and Wearing Apparel	5	49	27–29	4–5
5	Wood and Paper	6	50–55	30–31	6–7
6	Petroleum, Chemical and Non-Metallic Mineral Products	7	56–71	32–34	8–11
7	Metal Products	8	72–85	35–37	12
8	Electrical and Machinery	9	86–90	40–41	13–14
9	Transport Equipment	10	91–92	38–39	15
10	Other Manufacturing and Recycling	11–12	93–95	42	16
11	Electricity, Gas and Water	13	96–112	43–45	17
12	Construction	14	113–114	46	18
13	Trade	15–18	115–119	47	19–22
14	Transport	19	120–126	48–50	23–26
15	Post and Telecommunications	20	127	51	27
16	Financial Intermediation and Business Activities	21	128–135	52 - 54, 57	28–30
17	Public Administration, Education, Health and Other Services	22–24	136–162	55–56	31–35
18	Others	25–26	163	–	–

regions covered more than 90 percent of global GDP. The MRIO database was created from those SUT by resorting to several assumptions and auxiliary data, e.g. from the United States Geological Survey (USGS) and the Food and Agriculture Organization of the United Nations (FAO). The missing countries were covered in 5 additional "Rest of the World" regions. EXIOBASE does not provide time series but rather data for reference years. Version 1 was based on data from 2000, version 2 on data from 2007. A third version of EXIOBASE was published in 2018 that additionally contains time series from 1995 to 2011 (Stadler et al., 2018; Wood et al., 2018).

2.1.3. GTAP

The GTAP database v9 (Aguiar et al., 2016; Narayanan et al., 2012) was compiled for 140 regions and 57 industry sectors covering the years 2004, 2007 and 2011. Like EXIOBASE it does not yet cover time series. The database was created by relying on official data sources like the other three MRIO databases and additionally on their own network members who can submit SUT for their countries directly to GTAP. This way they gathered data covering more than 98 percent of global GDP. The disaggregation of SUTs was done with the help of auxiliary data, e.g. from FAO for agricultural data. By linking country input-output-tables (IOT) with the GTAP trade data, the MRIO database was created. Data gaps were filled by creating representative IOT for different regions from the existing IOT close to those regions.

2.1.4. WIOD

In contrast to the previous three MRIO databases that aimed at capturing environmental impacts of production activities in the world, the WIOD database (Timmer et al., 2015) was created with economic trade analysis in mind. It covers 41 regions and 35 industry sectors over a time series from 1995 to 2011. The 40 countries contained in the database account for more than 85 percent of global GDP. The rest was estimated in an additional "Rest of the World" region. The MRIO database was constructed by relying on only publicly available data, predominantly SUTs that were first harmonized into product-by-industry tables. From these, industry-by-industry IOTs were constructed with discrepancies being assigned to "Rest of the World". The time series had to be inferred from differently spread benchmark year tables of the national SUT.

2.2. SC analysis

As MRIO databases capture the economic links between industry sectors and regions, they can be used to analyze the amount and structure of supply (intermediate consumption) that is needed to meet a certain final demand. This can be done at an aggregated level (Leontief Inverse) or at a more detailed level (structural path analysis). These two ways are briefly introduced in this section.

In this research, the cumulated intermediate consumption (CIC) denotes the sum of all intermediate consumption that is required over all production tiers to meet a given demand. The CIC that is needed to meet the demand $\vec{f}$ can be calculated by applying the Leontief Inverse (Waugh, 1950):

$$\vec{v} = (I - A)^{-1} \cdot \vec{f} \tag{1}$$

$\vec{v}$ = vector containing total output (end product + intermediate consumption) by each industry sector for producing a defined unit of final demand

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[1] The aggregation logic derived coincided almost completely with the common classification system of Steen-Olsen et al. (2014) with the only exception that for this research it was decided to keep the "Others" sector for Eora and EXIOBASE as it could not be determined which other sectors to aggregate it with to keep consistency with GTAP and WIOD.

I = identity matrix
A = input coefficient matrix
$\overrightarrow{f}$ = final demand vector

The term $(I - A)^{-1}$ of this formula is called the Leontief inverse. If the final demand vector is set to retrieve the output of 1 USD of a specific industry sector in a specific region, the result in vector ($\overrightarrow{v}$) is the cumulated intermediate consumption needed to produce this output. Thus, only a distribution of the required intermediate consumption by regions and industry sectors is given but not a distribution by SC tiers. Structural path analysis (SPA) can be applied to gain more detailed insight into the distribution of intermediate demand by production tier.

SPA was introduced by Crama et al. (1984) and Defourny and Thorbecke (1984) and is also called the “Production Layer Decomposition” (Lenzen and Crawford, 2009). Equation (1) can be approximated by the power series of the Leontief Inverse to “unravel” (Suh and Heijungs, 2007) the intermediate consumption by production tiers (Lenzen and Crawford, 2009; Suh and Heijungs, 2007; Waugh, 1950; Wood and Lenzen, 2009):

$$\begin{aligned}(I-A)^{-1}\cdot\overrightarrow{f} &\approx A^0\cdot\overrightarrow{f} + A^1\cdot\overrightarrow{f} + A^2\cdot\overrightarrow{f} + \dots + A^n\cdot\overrightarrow{f} \\ &= I\cdot\overrightarrow{f} + A\cdot\overrightarrow{f} + A^2\cdot\overrightarrow{f} + \dots + A^n\cdot\overrightarrow{f} \qquad (2)\end{aligned}$$

n = number of desired production tiers.

Each exponential term (sometimes called power term) corresponds to a production tier and can thus be used in SPA to identify the main contributing industry sectors and the corresponding tiers (Lenzen, 2002; Wood and Lenzen, 2009).

SC analysis in general and SPA in particular have been used to complement life cycle inventory (LCI) compilation of Life Cycle Analysis (LCA). As LCA often operates with cut-off criteria to determine system boundaries, truncation errors can occur as the cut-off up- and down-stream can account for as much environmental impact as the analyzed processes (Lenzen and Dey, 2000; Lenzen and Treloar, 2003, 2002; Suh et al., 2004). By applying SPA to identify the main contributors of the upstream SC, the so-called hybrid LCAs can be carried out with the system boundaries drawn to include the relevant processes (Suh et al., 2004). It has been shown that regular LCA might underestimate embodied greenhouse gas (GHG) emissions in production chains by factor 2 if the processes for analyses were selected by cut-off criteria rather than by the identification of the main suppliers via structural path analysis (Lenzen and Treloar, 2002).

3. Methods

3.1. Comparison of MRIO databases

3.1.1. General comparison

The first step of comparing the chosen MRIO databases consisted of evaluating key parameters that reflected their overall structure. The following key parameters were chosen:

a) *Intermediate consumption*: For this, the sum of all transactions within the intermediate consumption matrix was taken.
b) *Value added*: Here, the sum of the value added across all regions and industry sectors was calculated.
c) *Total transaction volume*: The total transaction volume was defined as the sum of *intermediate consumption* (a) and *value added* (b).
d) *Share of intra-regional transactions*: To determine the share of intra-regional transactions, first the sum of all intra-regional transactions was calculated and then divided by the total volume of *intermediate consumption* (a).

The results of this comparison for the base year 2007 are presented in Table 4 in section 4.1.

3.1.2. Comparison of transaction structure of different MRIO databases based on unified dimensions

To enable comparability of the transaction flows captured within each database, several aspects had to be considered. First, a base year had to be defined. As the version of EXIOBASE that was used in this research was based on 2007 data, the tables of the same year were taken for the other three. Secondly, the structure of the MRIO databases were evaluated more closely. As they had very different dimensions, i.e. different granularity in the number of regions and industry sectors (c.f. Table 1), the MRIO tables had to be converted to a unified form on which basis they could be compared. The smallest common denominator was searched for in the number of distinct regions and industry sectors. This is commonly referred to as common classification (CC) system (Owen et al., 2016, 2014; Steen-Olsen et al., 2014).

- **Regions**: WIOD had the smallest number of regions and thus its list of regions was the smallest common denominator. So all regions that were not explicitly listed in WIOD were aggregated under “Rest of the World (RoW)” in the other MRIO databases. This left the number of regions in the unified form at 41.
- **Industry sectors**: Eora was the smallest common denominator regarding industry sectors as it had the lowest industry sector count (26) out of the chosen four databases. However, as some of the industry sectors in Eora had to be aggregated to be comparable to GTAP or WIOD this number was reduced further. For example, sector 13 (Trade) in Table 2 only had one corresponding industry sector in GTAP. In Eora, this sector was still disaggregated into “Wholesale Trade”, “Retail Trade”, “Maintenance and Repair” and “Hotels and Restaurants” (sectors 15–18). Therefore, these four industry sectors had to be aggregated leaving Eora with 22 distinct industry sectors instead of the original 26. Table 2 shows an overview of the aggregated industry sectors. In the end, 18 unified industry sectors could be identified. More information including a more detailed list of the original industry sectors can be found in the supplementary material (SM2).

The numbers listed under Eora, EXIOBASE, GTAP and WIOD represent the index of their industry sectors that were aggregated into the respective industry sector. For example, when looking at the unified industry sector “Textiles and Wearing Apparel”, for Eora no aggregation had to be performed as it corresponded to its 5th industry sector (“Textiles and Wearing Apparel”). For WIOD, however, its 4th and 5th industry sector (“Textiles and Textile Products” and “Leather, Leather and Footwear”) had to be aggregated. A more detailed overview of the aggregation logic is given in the supplementary material.

This process of bringing the MRIO databases to the same dimensions led to a final **unified form** of 41 regions and 18 industry sectors. 41 regions multiplied by 18 industry sectors resulted in 738 rows and columns for the intermediate consumption matrix. One row was added for the value added of each industry sector. The unified form was therefore a matrix with 739 rows and 738 columns containing 545,382 cells. Afterwards, the unified tables were converted to the coefficient form. This was done by dividing each transaction amount (i.e. cell value) by the production value (i.e. the column sum of the respective receiving sector). The coefficient form was used to allow for a simple evaluation of similarities and differences in the transaction structures of the MRIO databases.

The relative difference $\Delta_{A,B_{ij}}$ of every MRIO cell was determined using the following formula:

$$\Delta_{A,B_{ij}} = \begin{cases} 0, & A_{A_{ij}} = 0 \wedge A_{B_{ij}} = 0 \\ 3*\dfrac{\left|A_{A_{ij}} - A_{B_{ij}}\right|}{\left|A_{A_{ij}} + A_{B_{ij}}\right|}, & A_{A_{ij}} \neq 0 \vee A_{B_{ij}} \neq 0 \end{cases} \tag{3}$$

$A_{A_{ij}}$ = coefficient in $\text{cell}_{i,j}$ in the input coefficient matrix of MRIO database A
$A_{B_{ij}}$ = coefficient $\text{cell}_{i,j}$ in the input coefficient matrix of MRIO database B
$\text{cell}_{i,j}$ = cell in row i and column j

In this way, the relative difference of two equal values would result in 0 percent. If one value was double the size of the other, a relative difference of 100 percent would be indicated and in the case of one value being zero, the maximum difference of 300 percent would be the result of the formula. The relative difference was calculated for every cell in the unified form, i.e. for 545,382 cells. In order to visualize those differences the cells of the pairwise comparison were colored according to the calculated difference. The results of this calculation and visualization are presented in section 4.2. The relative differences were characterized in five classes as shown in Table 3. The threshold values were determined by the authors to classify the range of relative differences. Class 1 was assigned the darkest and class 5 the lightest shade.

For a summarized evaluation of the difference or similarity between MRIO databases, a range of comparison metrics exists (Gallego and Lenzen, 2009; Steen-Olsen et al., 2016; Wiebe and Lenzen, 2016). Wiebe and Lenzen (2016) narrowed the applicable matrix comparison metrics down as they were expecting their tables to also contain some values of zero. As this was also the case with this research, the selected list of Wiebe and Lenzen was taken as starting point for selecting an appropriate metric.

As more than one metric should be used to compare matrices (Gallego and Lenzen, 2009), the mean absolute distance (MAD) and the root mean square error (RMSE) were chosen for this research as they were most readily applicable and already used in other research, e.g. by Wiebe and Lenzen (2016) or Steen-Olsen et al. (2016). The formula for the MAD is shown in equation (4):

$$MAD = 100*\frac{\sum_{j=1}^{m}\sum_{i=1}^{n}\left|A_{A_{ij}} - A_{B_{ij}}\right|}{m \times n} \tag{4}$$

Table 3
Classification of relative differences and levels of similarities between MRIO databases. Class 1 represents the “best” category with relative differences not exceeding 0.6, whereas class 5 constitutes the “worst” category with relative differences of 2.4 or more.

Class	Range of relative differences	Level of similarity
1	$\Delta < 0.6$	Very High
2	$0.6 \leq \Delta < 1.2$	High
3	$1.2 \leq \Delta < 1.8$	Medium
4	$1.8 \leq \Delta < 2.4$	Low
5	$\Delta \geq 2.4$	Very Low

$A_{A_{ij}}$ = coefficient in $\text{cell}_{i,j}$ in the input coefficient matrix of MRIO database A
$A_{B_{ij}}$ = coefficient in $\text{cell}_{i,j}$ in the input coefficient matrix of MRIO database B
$\text{cell}_{i,j}$ = cell in row i and column j
n = number of rows
m = number of columns

In this study, the MAD was slightly altered to express the difference between two matrices normalized to the maximum possible difference between two MRIO tables in the coefficient format. As the maximum possible result of the term $\sum_{j=1}^{m}\sum_{i=1}^{n}\left|A_{A_{ij}} - A_{B_{ij}}\right|$ is equal to 2 × m, the following holds true:

$$0 \leq \frac{\sum_{j=1}^{m}\sum_{i=1}^{n}\left|A_{A_{ij}} - A_{B_{ij}}\right|}{2 \times m} \leq 1 \tag{5}$$

Therefore, the modified MAD (MAD_mod) is calculated according to equation (6):

$$MAD\_mod_{A,B} = 100*\frac{\sum_{j=1}^{m}\sum_{i=1}^{n}\left|A_{A_{ij}} - A_{B_{ij}}\right|}{2 \times m} \tag{6}$$

Two databases with perfect conformity would lead to an MAD_mod of 0, whereas total discordance would result in a score of 100. The results of the MAD_mod and the RMSE for the summarized comparison of MRIO databases can be found in section 4.2.

3.2. Comparison of SC analyses of different MRIO databases based on original form

To compare the existing databases regarding the consistency of SC analyses, this second part of the analysis used the coefficient form of the MRIO databases. Here, the unified MRIO database form as described in section 3.1.2 was not used but the original coefficient form of the databases as a user also would apply the database in its original form. The analysis was done by defining a common starting point, i.e. tier 1 supplier. For this analysis, the SC of a metal sheet supplier in Germany was chosen.

The following sectors were chosen to represent the metal sheet supplier in Germany:

- Eora: “Metal Products”
- EXIOBASE: “Manufacture of basic iron & steel and of ferro-alloys and first products thereof”
- GTAP: “Ferrous Metals” and “Metal Products”
- WIOD: “Basic Metals and Fabricated Metal”

To evaluate the robustness of the analysis the effect of sector reallocation (switch of the initial industry sector) or insecurity in the assignment of a product to a certain industry sector, the sectors “Ferrous Metals” and “Metal Products” of GTAP were included in the analysis. This allowed for evaluating effect it had on the SC analysis outcome if, e.g. one analyst would have chosen to represent a metal sheet supplier by the “Ferrous Metals” industry sector in GTAP while another one would have chosen the “Metal Products” industry sector instead. If not otherwise declared in the context of the SC analysis, GTAP refers in this paper to GTAP (Ferrous Metals).

3.2.1. SC analysis based on the CIC

The Leontief Inverse (c.f. section 2.2) was used to determine the CIC needed to produce an output of 1 USD of the chosen industry sector (see above). To enable comparability the intermediate consumption was afterwards aggregated according to the CC (section 3.1.2). To evaluate the similarity of this first way of SC analysis between the four databases, the MAD of the CIC by industry sector was calculated using its unaltered formula (c.f. equation (4)). To enable a more detailed comparison between databases, the top 10 supplying regions and industry sectors by CIC were identified as they should indicate were most of the production activities are located. The results of the first way to approach SC analysis are presented in section 4.3.1.

3.2.2. *SC analysis by production tiers using SPA*

The second way of approaching SC analysis was done by disaggregating the intermediate consumption to show the SC tiers with the help of equation (2) (c.f. section 2.2). The focus was laid on identifying the top suppliers in tier 2 and tier 3 of the chosen tier 1 supplier. It was defined that the 10 top sub-suppliers for tier 1 should be identified (tier 2) by taking the 10 supplying industry sectors in the MRIO database with the highest coefficients, i.e. highest transaction flows. For instance, if the second largest coefficient for the chemical sector in Germany was found to be the chemicals sector in the United States, a theoretical chemicals supplier in the US was assumed to be a tier 2 supplier. For tier 3, the number of suppliers was limited to 30 in total. That means that for every tier 2 supplier, 3 sub-suppliers (tier 3) were identified using the same logic as for identifying tier 2 suppliers. The analysis yielded 40 fictional suppliers for tier 1 (10 × tier 2 + 30 × tier 3) that were defined by industry sector and region. Therefore, the approach applied in this research was a modified SPA as the number of suppliers for each production tier was limited. The approach is visualized in Fig. 1. Starting from this structure, it was determined how many matches, i.e. congruent region and industry sector combinations, between SC analyses of the different MRIO databases existed.

For comparing two MRIO databases in a condensed way, a “similarity score” was defined. The results of the SC analysis, i.e. the sector and region identified as suppliers, were compared pairwise between the MRIO databases. It was evaluated whether the resulting tier 2 and tier 3 suppliers of database A matched with the ones of database B. For example, assume that in Fig. 1 the first supplier in tier 3 of MRIO analysis using database A was located in China (“Region A.3.1”) and operating in the industry sector “mining and quarrying”. While the fourth supplier in tier 3 of the MRIO analysis using database B was also located in China (“Region B.3.4”) and operating in the “coal” sector. With the two regions being the same, the industry sectors also have to be the same to be counted as a match. To determine whether similar industry sectors like the ones in the example could be counted as match, the aggregation logic of Table 2 was used. As “mining and quarrying” of database A and “coal” of database B fall into the unified industry sector “Mining and Quarrying”, the two suppliers were counted as a match. The position of the suppliers was given no weight which meant that if supplier A.2.3 and supplier B.2.1 in Fig. 1 were of the same region and industry sector, they would have been counted as a match. If all 10 tier 2 suppliers of MRIO A were congruent to MRIO B, it would result in 10 matches for tier 2. The same logic was applied to tier 3. The total number of possible matches was 10 for tier 2 and 30 for tier 3, thus 40 in total.

The similarity score $L_{A,B}$ to express the similarity of SC analyses by databases A and B was calculated by the following formula:

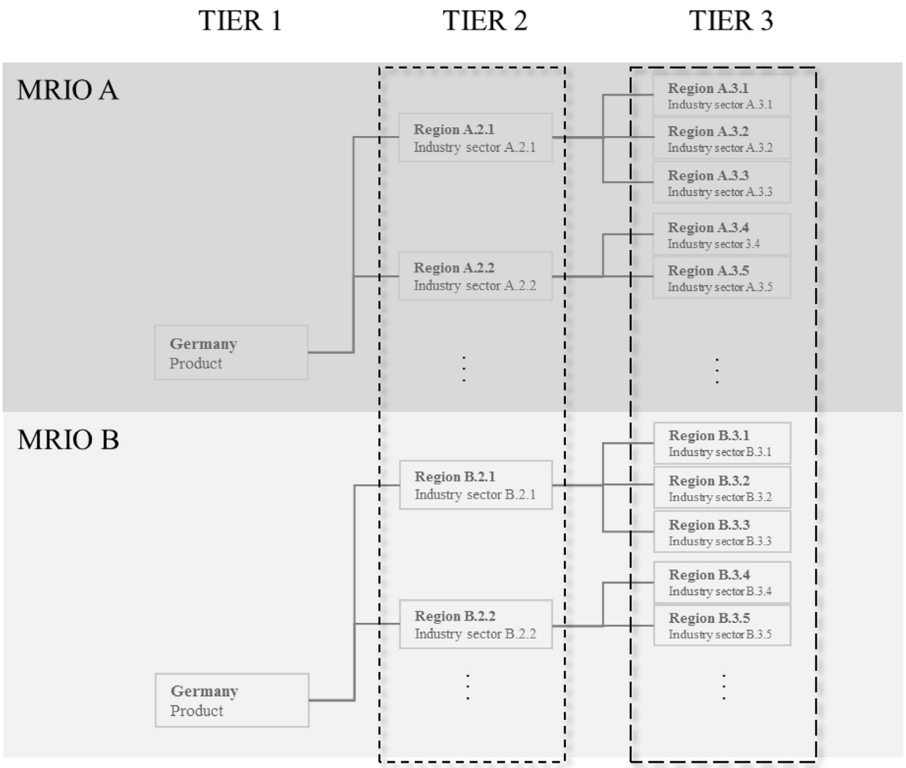


Fig. 1. Structure of the simulated SC used for comparing MRIO databases.

$$L_{A,B} = \frac{\textit{number of matches between A and B}}{\textit{number of possible matches}} \tag{7}$$

This means that a similarity score of 1 would denote 100 percent of conformity between SC analyses, i.e. the sectors and regions of all 40 suppliers were the same. A score of 0 would indicate entirely different results of the SC analyses with none of the 40 suppliers being the same in terms of region and/or sector. For example, if five suppliers in tier 1 were a match between the SC analyses of MRIO database A and B and seven suppliers in tier 3 were a match, the $L_{A,B}$ would result in $(5 + 7)/40 = 0.3$. The results of the second way to approach SC analysis are presented in section 4.3.2.

3.3. Comparing the SC analysis results with the real SC

To check the theoretical MRIO-based SC analyses as retrieved by the two ways of approaching SC analysis (as described in section 3.2.1 and 3.2.2) for the four databases against a real SC, a case study was carried out to gather primary data about the SC structure of a metal sheet supplier in Germany. The process of selecting case studies followed two considerations. On the one hand, they had to be a relevant example for an industrial company and on the other hand they should have a decent chance of success. As generating transparency in the SC always touches on confidentiality agreements between SC companies, the chances of success were determined by the already existing transparency and the type of relation with the immediate suppliers and sub-suppliers. This research carried out a case study investigating the SC of a metal sheet supplier. A data collection form was sent via Email to the tier 1 supplier first, where it was asked to enter all its suppliers, their industry sector and the location of their production as well as a respective contact persons. After the tier 2 suppliers were identified, they were approached and asked to fill out the same data collection form to identify their suppliers, i.e. the tier 3 suppliers. To compare how close the results of a SC analysis was to the real SC, the matches of the SC analyses with the real SC were evaluated. The same similarity score and method for identifying matches was applied as for the comparison of the SC analyses to each other (c.f. section 3.2.2). As the second way to approach SC analysis yielded 40 theoretical suppliers (c.f. section 3.2.2), the top 40 supplying industry sectors by amount in USD were also chosen for the first way (based on the CIC). The case study was intended to give an indication on how SC analyses captured a real SC.

To compare a single SC of a case study against the results of an average SC derived from an MRIO might seem as a provocative approach and statement. However, the wish and tendency to gain particular insight into SC structures for specific companies is spawning the use of MRIO for particular SC analyses and social hotspot databases (Benoit-Norris et al., 2012; Ciroth and Eisfeldt, 2015; Systain, 2018; Zimmer et al., 2017). As the secondary motivation of this research was to evaluate the possible application of MRIO based SC analysis for social hotspot assessment, the comparison of SC analysis results against a real SC was included.

4. Results and discussion

4.1. General comparison

The key parameters of the evaluated MRIO databases are shown in Table 4.

The total transaction volumes of all four MRIO databases were in the same order of magnitude with an average total of 1.10×10^{14} USD from which the strongest deviation was +3.3 percent (Eora) and −2.9 percent (EXIOBASE). These results show that the evaluated databases arrived at a similar grand total of transaction volumes of the world's economy. The slightly increased volume of Eora could be due to the way the database was constructed as their approach was to incorporate all available sources (c.f. section 2.1.2) claiming that it covered virtually 100 percent of the global GDP. The relatively low transaction volume of EXIOBASE might be caused by the currency conversion from EUR to USD. The conversion was done using an average exchange rate for 2007 (1.3705 USD/EUR) from the European Central Bank. As conversion rates are dynamic there is an inherent inaccuracy in applying an average exchange rate to the transaction volume of an entire year. In 2007, the exchange rate had a peak value of 1.4875 USD/EUR. Using this peak exchange rate, EXIOBASE would show a total transaction volume of 1.16×10^{14} USD and range above Eora. Thus, it was not possible to determine with certainty how EXIOBASE compared to the other databases in terms of captured transaction volume.

The share of intra-regional transactions in the intermediate consumption was more than 80 percent for all databases. This clearly shows that the majority of transactions captured in MRIO databases was between industry sectors in the same region. Inter-regional transactions therefore only accounted for about 20 percent of the transactions. This seems surprisingly low as trade globally has a share of about 50–60 percent of GDP (The World Bank 2017) and to capture the multi-regionality of trade is the main target of MRIO databases. Even though the comparison of the inter-regional transaction share in MRIO databases and the data provided by the World Bank is not looking at the exact same information, their disparity cannot be fully explained as the construction method of MRIO databases constructs the inter-regional transactions based on trade balances and national GDP (c.f. section 2.1). The effect seems to be systemic because all four databases showed this strong weight in intra-regional transactions. It would be interesting for future research to look more closely at how the intra- and inter-regional transaction volumes are constructed in relation to GDP and how this relation of 80 to 20 percent is generated.

In conclusion, the overall macro-level key parameters of the databases show a surprisingly similar range, but they are dominated by intra-regional transactions. Future research should look at where this strong weight of intra-regional transactions results from.

4.2. Comparison of transaction structure

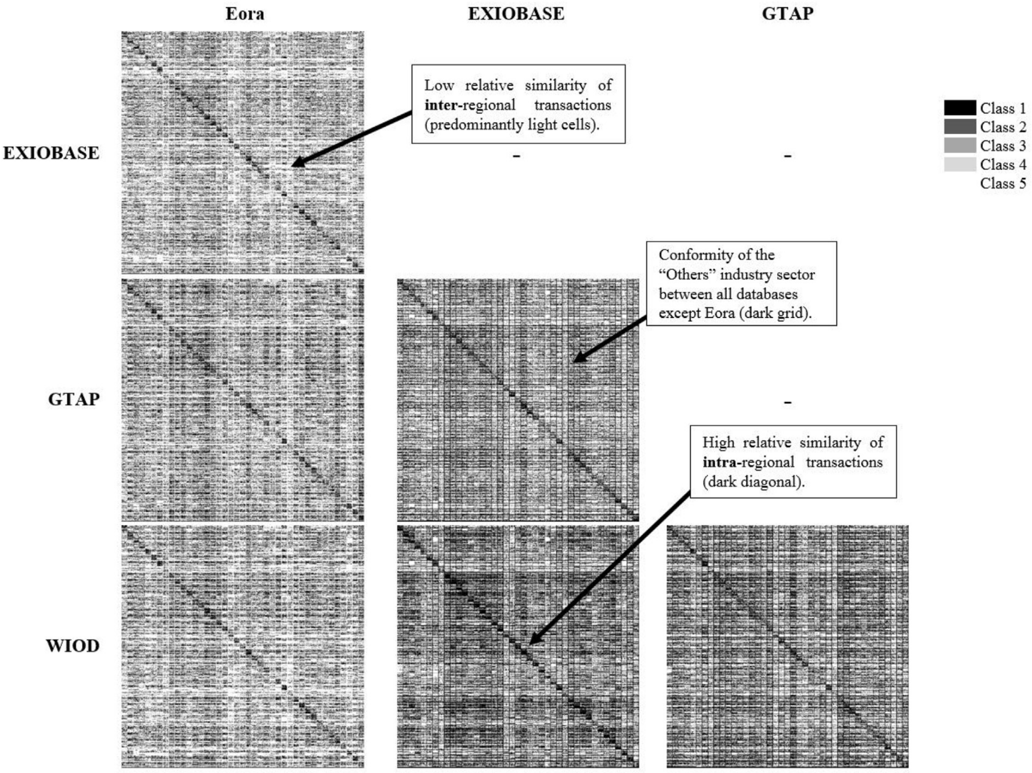
In addition to the key parameters discussed in section 4.1, Fig. 2 shows the visualized relative differences of the pairwise comparisons of sectors and regions in the four databases considered. Darker shades indicate higher similarity between MRIO databases. It can be seen by the clearly discernible diagonal in every picture that intra-regional transactions were the most similar between all four databases. All databases showed clear differences between each other when it came to transactions between regions. The high similarity in intra-regional transactions could be explained on the one hand that they were the largest transaction volumes in the databases and thus absolute differences of intra-regional transfers did not weigh in as much on the overall transaction volume. On the other hand, intra-regional transactions are captured in SUT that all MRIO databases used as base for their model construction. The way the inter-regional transactions were calculated differ between databases (c.f. section 2.1). This leads to the assumption that the premises of MRIO database construction are the deciding factor in the way the transaction flows are determined within each database.

The MAD_mod expresses the relative differences in transaction structure in combination with transaction volume (c.f. section 3.1.2). The MAD_mod and RMSE values in Table 5 show in a condensed way that even though all MRIO databases differed

Table 4
Overview of the MRIO database key parameters.

	Reference year	Intermediate Consumption [USD]	Value Added [USD]	Total transaction volume [USD]	Share of intra-regional transactions
Eora	2007	6.05×10^{13}	5.32×10^{13}	1.14×10^{14}	81.2%
EXIOBASE[a]	2007	5.48×10^{13}	5.21×10^{13}	1.07×10^{14}	84.6%
GTAP	2007	5.64×10^{13}	5.39×10^{13}	1.10×10^{14}	83.2%
WIOD	2007	5.50×10^{13}	5.44×10^{13}	1.09×10^{14}	81.7%

[a] Values in EXIOBASE were originally in EUR and were converted to USD by using the European Central Bank conversion factor of 2007.



Low relative similarity of **inter**-regional transactions (predominantly light cells).

Conformity of the “Others” industry sector between all databases except Eora (dark grid).

High relative similarity of **intra**-regional transactions (dark diagonal).

Fig. 2. Relative differences of MRIO databases identified by pairwise MRIO comparison. Relative differences were sorted into five groups: Group 1 = relative difference <60%, Group 2 = relative difference <120%, Group 3 = relative difference <180%, Group 4 = relative difference <240%, Group 5 = relative difference ≤300%. Shades range from dark (Group 1) to light (Group 5).

Table 5
MAD_mod and RMSE of pairwise MRIO database comparison based on transaction structure. The left figure shows the MAD_mod where a score of 0 denotes perfect conformity between databases, whereas a score of 100 would denote total discordance. The right figure shows the RMSE where a lower score shows higher similarity between databases.

	Eora	EXIOBASE	GTAP
	MAD_mod | RMSE	MAD_mod | RMSE	MAD_mod | RMSE
EXIOBASE	23.4 | 11.12×10^{-6}	–	–
GTAP	24.2 | 12.19×10^{-6}	19.1 | 8.10×10^{-6}	–
WIOD	22.2 | 10.93×10^{-6}	13.3 | 4.98×10^{-6}	17.7 | 7.64×10^{-6}

clearly from one another in their inter-regional transaction flows (c.f. Fig. 2), the overall differences were quite low with no pairwise comparison showing MAD_mod values above 24.2 and RMSE ranging within the same order of magnitude.

These high similarities between MRIO databases expressed by two metrics underscore again the weight of the intra-regional transactions. The significant differences of inter-regional transactions did not influence the scores so much as they only accounted for around 20 percent of the entire intermediate consumption volume (c.f. Table 4). Thus, the large and relatively similar intra-regional transactions led to an overall high similarity of MRIO databases. Eora differed the most from the other three databases (MAD_mod scores from 24.2 to 22.2 and RMSE between 1.09×10^{-5} – 1.22×10^{-5}) while EXIOBASE and WIOD were most similar to one another out of the given set (lowest MAD_mod score of 13.3 and lowest RMSE of 4.98×10^{-6}). This underpins that the new approach by Eora to construct an MRIO database led to a more different structure compared to the other databases. The high similarity between WIOD and EXIOBASE might lie in the fact that their construction was also very similar (focus on accounts for less regions but accounts from strong economies) while GTAP used an additional data acquisition method by resorting to members of their network. These findings agree with the research of Owen et al. (2016, 2014) who also found that inter-regional transactions were different between databases due to their different construction processes but overall conformity was high. This research adds to the findings by including EXIOBASE – in comparison to Owen et al. (2016, 2014) – and by additionally expressing the differences of the overall transaction structures between databases in a summarized way.

The results based on aggregated MRIO databases may have come with some distortion due to the aggregation process. Steen-Olsen et al. (2014) found that the initial level of detail before aggregation in MRIO analysis led to distorted results when looking at CO_2 multipliers for different industry sectors as the level of detail of the original information varied across databases. They did, however, not find the variability of the economic structure to be as high as the environmental extension data. With regard to differences in industry sector outputs Miller and Shao (1990) also found that sectoral aggregation led to higher inaccuracies than regional aggregation. Other studies also found that the aggregation of sectors and/or regions of MRIO databases led to changes in CO_2 accounts (Lenzen et al., 2004; Wyckoff and Roop, 1994) and that disaggregation is preferable to aggregation (Lenzen, 2011). Su et al. (2010) and Bouwmeester and Oosterhaven (2013) on the other hand found that few major industry sectors were sufficient to capture the majority of embodied CO_2 emissions in a product. Future comparison approaches should therefore look to find ways to compare the disaggregated MRIO data to compare the strengths of the individual MRIO construction approaches.

In summary, the results of the analysis of the four chosen MRIO databases showed significant differences in inter-regional transactions but overall a decent similarity between databases as the transaction volume was dominated by similar intra-regional transfers.

4.3. Comparison based on SC analyses

4.3.1. SC analysis based on the CIC

Table 6 shows the original MAD (c.f. equation (4)) for the similarity between the distributions of the CIC by industry sector between MRIO databases.

They showed a similar picture to the comparison of the transaction structure (Table 5) with Eora being the least similar to the rest (MAD > 0.179) and EXIOBASE and WIOD being the most similar (MAD = 0.075). Even though the comparison between both GTAP analyses had the lowest MAD (0.00018), sector reallocation showed a clear effect, as the similarity between GTAP (Metal Products) to WIOD was clearly higher than of GTAP (Ferrous Metals) to WIOD (MAD of 0.089 vs. 0.158). The similarity to EXIOBASE did not change significantly through sector reallocation, which might be because the respective sector in EXIOBASE comprised the manufacture of ferrous metals as well as of products thereof. Thus, it would be expected that GTAP (Metal Products) would be clearly more similar to Eora, which was represented by its “Metal Products” sector, than GTAP (Ferrous Metals). This was the case but with a change from 0.273 to 0.209 in the MAD value, the effect of sector reallocation was not as clear as between GTAP and WIOD. This might be due to the different approach to constructing Eora as compared to the other MRIO databases. Fig. 3 shows the top supplying countries and industry sectors of the CIC.

Fig. 3 clearly shows that Germany was the main supplying country across all databases with a share of supply ranging between 62% and 73%, followed by France, the Rest of the World, Italy, and China. Furthermore, the top 10 supplying countries constituted the majority of supplying countries with a total share of 83%–91% of the entire CIC across databases. The picture was not so consistent across databases when the distribution by industry sector was concerned. In all MRIO analyses, the tier 1 supplying industry sector (metal sheets) was the major supplying industry sector with a share of supply ranging from 47% (EXIOBASE) to 64% (Eora). The second largest supplying industry sector was also rather similar across databases referring to “Business Services” and “Other Business Activities”. Interestingly, the mining sector was not included in the top 10 supplying industry sectors in the analyses from GTAP (Metal Products) and WIOD.

The effect of sector reallocation could also be seen in the composition and ranking of the top 10 supplying industry sectors of GTAP (Ferrous Metals) and GTAP (Metal Products), respectively. However, the distribution pattern appeared very similar with 9 out of the 10 top industry sectors being the same just in different order. Only the “Petroleum, Coal Products” sector of GTAP (Ferrous Metals) was replaced in the top 10 of GTAP (Metal Products) by “Chemical, Rubber, Plastic Products”. The difference between the structures of GTAP after sector reallocation were less obvious in total than between GTAP and EXIOBASE. Similarly to the distribution by countries, the top 10 industry sectors also covered the majority of the CIC with total shares ranging from 84% to 94%. An exception here was EXIOBASE where only roughly two thirds of CIC were covered by the top 10 supplying industry sectors. This might be caused by the comparatively high sectoral resolution of EXIOBASE (163 industry sectors compared to 26–57 industry sectors in the other MRIO databases, c.f. Table 1) that would enable a more detailed distribution of intermediate consumption between industry sectors.

These results show that basic information on main SC structures can be deduced from an MRIO analysis applying the Leontief Inverse. However, while not delivering contradicting results, the difference in the distribution of CIC over regions and industry sectors between MRIO databases is still apparent meaning that the choice of MRIO database influences the results of SC analysis. Robustness regarding sector reallocation was less an influencing factor than MRIO database construction itself.

4.3.2. SC analysis by production tiers

The similarity scores based on section 3.2.2 quantified the similarities/differences between the databases based on the matches of the SC analyses. The total possible number of matches was 40. The calculated similarity scores are shown in Table 7. The percentages indicate the overall matches of region and industry sector between the databases and the number in brackets indicates the number of matching tier 1 and tier 2 suppliers, respectively. As an example, the comparison of the same SC analysis of WIOD and

Table 6
Mean absolute distance (MAD) of the CIC for the production of 1 USD output by industry sector.

	Eora	EXIOBASE	GTAP (Ferrous Metals | Metal Products)
EXIOBASE	0.179	–	–
GTAP (Ferrous Metals | Metal Products)	0.273 | 0.209	0.150 | 0.141	0.000
WIOD	0.182	0.075	0.158 | 0.089

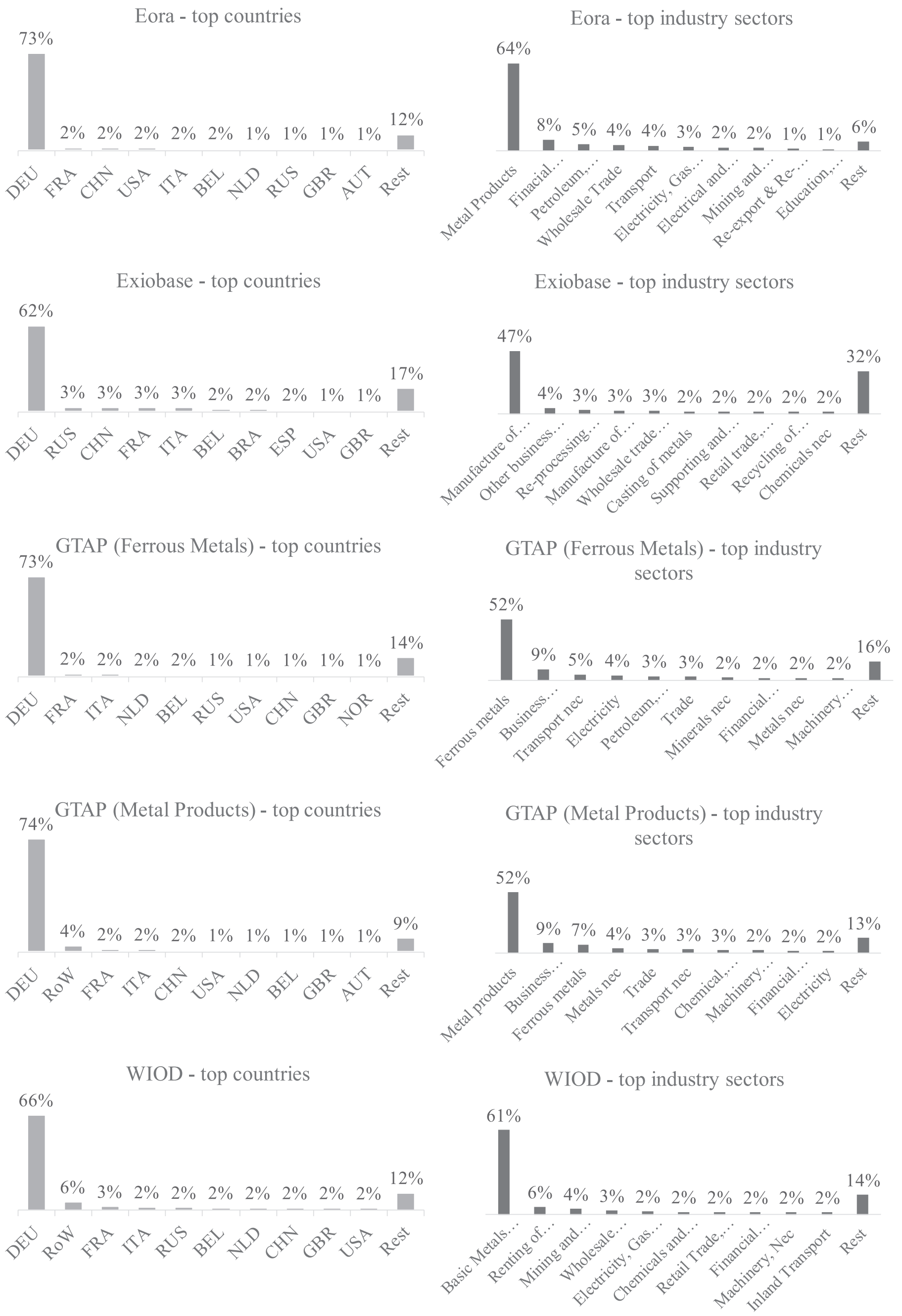


Fig. 3. Top supplying countries and industry sectors for a metal sheet supplier in Germany based on the cumulated intermediate consumption analysis.

GTAP revealed, that only 50 percent of the resulting SC did match (20 out of 40). Four matches occurred in tier 2 and 16 matches in tier 3.

It is apparent that the SC analyses differed more between one another as the MRIO databases did in the comparison based on transaction flows or CIC. The highest similarity could be found between Eora and GTAP as well as Eora and WIOD where more than 60 percent of the SC sections coincided. However, the lowest

Table 7
Similarity scores of pairwise MRIO database comparison based on SC analysis. The numbers of matches in tier 2 and tier 3 are given in brackets.

	Eora	EXIOBASE	GTAP
EXIOBASE	52.5% [4 + 17]	–	–
GTAP	62.5% [6 + 19]	52.5% [4 + 17]	–
WIOD	62.5% [6 + 19]	57.5% [6 + 17]	50.0% [4 + 16]

Table 8
Regional spread of the value added by the simulated suppliers for 1 USD of purchased final product in Germany.[2] Due to rounding, some totals may not correspond with the sum of separate values.

	Eora			EXIOBASE			GTAP			WIOD		
VA [USD]	0.552			0.600			0.557			0.537		
Region	Tier 1	Tier 2	Tier 3	Tier 1	Tier 2	Tier 3	Tier 1	Tier 2	Tier 3	Tier 1	Tier 2	Tier 3
Austria		0.5%	0.1%									
Belgium		0.4%	0.1%			0.1%					0.8%	0.1%
France		0.7%	0.3%		1.3%	0.3%					1.1%	0.4%
Germany	50.0%	32.5%	15.4%	72.8%	20.3%	4.2%	56.0%	34.1%	9.5%	65.3%	21.3%	4.9%
Italy					0.8%	0.2%					0.9%	0.2%
Netherlands												0.1%
Norway									0.2%			
Russia						0.0%			0.2%			
RoW											3.5%	1.4%
Total	50.0%	34.2%	15.8%	72.8%	22.4%	4.8%	56.0%	34.1%	9.9%	65.3%	27.6%	7.1%
# of regions	1	4	4	1	3	5	1	1	3	1	5	6

VA = Value Added.
RoW = Rest of the World.

similarity was identified between GTAP and WIOD with 50 percent conformity, placing Eora between GTAP and WIOD with similarities to both. This means that depending on the MRIO database chosen for SC analysis, a possible subsequent hotspot assessment would yield rather different potential hotspots than if another database had been chosen for the SC analysis.

The theoretical expectation for SC analysis would be that EXIOBASE and WIOD would yield the most similar SC, as their transaction structure was closest to each other (c.f. Table 5). However, their SCs were not the most similar. This leads to the assumption that the differences in the initial structure, i.e. before unification (c.f. Table 1), were the more determining factor in the different results. The small relative differences of intra-country transactions seem to be significant enough to yield different results.

To evaluate the influence of the strong intra-regional transactions, the regional spread of the analyzed SCs was evaluated as well. To get an idea where most of the value creation took place, the value added of the MRIO analysis was taken and added by country for every tier of the analyzed SC. Table 8 shows the regional spread of value added that the analyses yielded for the different MRIO databases.

As shown in Table 8, all MRIO databases had the majority of value added coming from tier 1 with descending shares to tier 3. GTAP exhibited the lowest regional spread with the entire value added in tier 2 coming from one region (Germany) and in tier 3 still mainly coming from Germany. WIOD showed the strongest regional spread with value added in tier 2 already coming from 5 and in tier 3 coming from 6 different regions. The weight of intra-country transactions can be seen again in this analysis as the majority of value added was coming from Germany (>90 percent) in all analyses (c.f. Table 8). When looking at the number of regions in Table 9 it can be seen that the regions, that the analyzed SCs covered over all three tiers, differed by MRIO database. In the analysis of GTAP, suppliers along the SC came from 5 different regions, of Eora from 9, of EXIOBASE from 9 and of WIOD from 12 different regions. Regarding the potential of MRIO analysis for enabling a hotspot assessment and therefore supporting the guiding of data collection efforts in S-LCA, this difference in identified number of regions showed an inconsistent and arbitrary result across MRIO databases. This shows that MRIO analyses do not provide consistent results that are independent from a chosen database. On the one hand, this might be a direct result of the different construction approaches of MRIO databases, reflecting the different degrees of data granularity and the quality of the sources they come from. For practitioners this would mean that they should carefully choose the MRIO database for their intended analysis and evaluate, whether the respective construction approach of the MRIO database could be beneficial or detrimental to the intended analysis. On the other hand, this raises the question, whether one MRIO database more accurately reflects the real transaction flows than the other three. For practitioners this would mean that the use of several MRIO databases is advised as an analysis result can be regarded as robust if several MRIO analyses arrive at the same result. Future research could try to combine the strengths from MRIO databases and produce a consistent synthesis of all of them.

It could be also expected that MRIO databases with a low number of industry sectors would have a larger spread of regions because there are not so many industry sectors available in every region. Furthermore, it could be expected that MRIO databases with many regions covered would also have a larger spread of regions than databases with less regions. Thus, it would have been an

Table 9
Number of matches of the SC analysis based on the CIC with the case study results.

	matches	%
Eora	1	4.8%
EXIOBASE	3	14.3%
GTAP (Ferrous Metals)	3	14.3%
GTAP (Metal Products)	0	0.0%
WIOD	2	9.5%

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[2] Table 8 shows the distribution of value added as yielded by the different SC analyses. The sum of values added by the identified suppliers was normalized to 100%. Suppliers of different industry sectors were aggregated when they were located in the same region. E.g. 56% of the value added in the SC analysis by GTAP (0.557 USD) happened in tier 1, Germany. In tier 3 of this analysis, only 9.9% of the value added were contributed where Germany still held the major share with 9.5%.

expected result that Eora (lowest industry sector count and highest region count) would be the MRIO database with the largest regional spread of the evaluated databases. However, EXIOBASE and WIOD showed a larger or similar spread compared to Eora. One could assume that the total transaction volume captured in an MRIO database would be an influencing factor on the regional spread as databases with more transactions captured might have a bias towards inter- or intra-regional transactions. But the two closest MRIO databases to the average transaction volume (GTAP and WIOD, c.f. Table 4) differed the most in their regional spread leading to the conclusion that the transaction volume is not the most influencing factor.

A possible explanation for the degree of regional spread could be again the premises under which the MRIO databases were created. WIOD, where the intra-regional transactions were the weakest (91.6 percent value added in Germany), was compiled with focus on economic trade analysis. GTAP, where the intra-regional transactions were the strongest (99.6 percent value added in Germany), was compiled by additionally relying on individually provided SUT from their network. This might have resulted in a stronger accuracy and volume of national accounts, strengthening the weight of intra-regional transactions. As the building process of MRIO databases could not be reconstructed entirely from the available publications, the influence of the construction process could not be examined in detail in this paper.

At this point, a critical view on the assumption of taking the value added as metric for assessing SC structures is adequate. For this research it was decided to take the indicator value added as it reflects economically strong supplying industry sectors which could thus be seen as major and stable parts of a SC. This was deemed sufficient as the first motivation was to determine the structure of the SC. However, the indicator value added might give misleading information when trying to identify socially critical SC steps. For instance, the step with the highest value added of a smartphone production might be the end producer which adds a margin on its product without putting significant additional labour into it. The step of smartphone assembly might not appear as significant in relation to the last step or even to the manufacturing of other components but house the possibility for higher social violations as the other production steps. Therefore, current social database providers (SHDB, PSILCA etc., c.f. section 1) are often translating the monetary units in MRIO databases into working hours by using information on average labour costs for the specific regions and industry sectors and combining them with the information on intermediate consumption (Alsamawi et al., 2014; Benoit-Norris et al., 2012; Ciroth and Eisfeldt, 2015; Simas et al., 2014).

In general, the results of the comparison of SC analyses underpinned the results obtained from the comparison based on transaction flows. MRIO were similar with regard to the intra-regional transactions but differ in their inter-regional transactions. When assessing the inter-regional differences in more detail, the differences between the databases were substantial but difficult to understand and interpret. The main reason for the differences between the databases is expected to be due to the different MRIO database creation processes.

The SC analysis by production tier gave a more detailed SC structure than the analysis of the cumulated intermediate consumption. The question whether the increase in detail constitutes more meaningful information on SC structures has no clear answer. The cumulated analysis (Leontief Inverse) gave a comprehensive but aggregated picture of the SC structure and it can be assumed that the main suppliers were contained in the resulting SC. As the more detailed approach using the disaggregated intermediate consumption (Structural Path Analysis) and the concentration on a limited number of main suppliers left out a large amount of other SC elements the more detailed SC structures derived have to be interpreted more cautiously. Especially when considering that all MRIO database data is afflicted with uncertainty, the error propagation that occurs with the calculation of more detailed SCs can have a significant impact on the results (Heijungs and Lenzen, 2014).

4.3.3. Comparison with real SC

A case study for the investigation of the SC structure of a metal sheet supplier in Germany was carried out (see section 3.3). In the course of the case study 10 tier 2 suppliers were identified by the initial metal sheet supplier. The subsequent tier 3 suppliers were identified through further inquiry and yielded 11 additional suppliers. The structure of the SC in the case study differed from the pre-defined structure of the simulated SC especially in tier 3 as only 11 sub-suppliers were identified instead of, as expected, at least 30. This was mainly due to the fact that the SC for some materials ended with tier 2. Tables 9 and 10 show the matches of the SC analyses to the case study SC.

The degree of overall matches between the real SC and the theoretical ones modelled with the four databases was rather low. This was not as surprising as the degree of accuracy decreases with every production layer due to the error propagation of the inherent uncertainties contained in the initial data set (Heijungs and Lenzen, 2014; Lenzen et al., 2012). Table 9 shows that of the SC analysis based on the CIC the SC structure of EXIOBASE and GTAP (Ferrous Metals) came closest to that of the case study with 14.3% of theoretical suppliers matching. The effect of sector reallocation also became apparent here as the GTAP (Ferrous Metals) analysis had a higher similarity to the case study SC than the GTAP (Metal Product) analysis indicating the differences in SC structures yielded by the different “starting sectors”.

The overall matches decreased when applying the SC analysis by production tier as can be seen in Table 10. The very weak, but still best matches of the WIOD analysis (14.3%) have to be taken with a pinch of salt as they occurred in the “Rest of the World” region. This means that it is not guaranteed that the country of the case study was actually matched in WIOD as a lot of countries are covered in WIOD's “Rest of the World” region. For example, one real supplier was the mining and quarrying sector in Kazakhstan. As this country was assigned to “Rest of the World” a match was registered, but it is not possible to determine what weight Kazakhstan held in the “Rest of the World” region in WIOD. GTAP's analysis yielded one match with the actual SC. This was rather a result of the sector aggregation logic as the analysis found a supplier in the mineral mining sector of Germany whereas the actual supplier in the case study was pelleting coal in Germany. As both activities were subsumed under the mining and quarrying sector, they were counted as a match. One of the main differences between MRIO analyses and case study was that all SC analyses identified strong supply from the same country (Germany) due to the inherent strong depiction of intra-regional transactions of MRIO databases (see section 4.2).

Keeping in mind that a single case study can only deliver an indication, a few observations could be made. The overall decrease

Table 10
Number of matches of the SC analysis by production tiers with the case study results.

	match tier 2	match tier 3	%
Eora	0	0	0.0%
EXIOBASE	0	0	0.0%
GTAP	1	0	9.5%
WIOD	1	2	14.3%

of matches between the SC analysis based on CIC (9 in Table 9) and the SC analysis by production tiers (4 in Table 10) indicated that the more aggregated view on the SC yielded higher conformity with the aggregated case study SC than the more detailed analysis. This is, on the one hand, due to the missing distinction between SC tiers when determining matches leading to higher overall matches. On the other hand, the SC analysis based on CIC also yielded another structure as all intermediate consumption of an industry sector in a country added up regardless of production tier. Therefore, some industry sectors made it into the top supplying industry sectors that were neglected in a focused analysis looking at production tiers, e.g. the "Mining & Quarrying" sector was more prominently identified by the aggregated analysis than by the analysis disaggregated by production tiers. This singular comparison seemed to give the indication that the more aggregated view on SCs yield more practical results as they better represent an actual SC and as the spurious accuracy of the more detailed analysis lead to less useable results. Furthermore, the required resolution for the analysis at such a detailed level is hindered by low database granularity and the heterogeneity of industry sectors within MRIO databases (Lenzen et al., 2012; Wiedmann et al., 2011). (Kucukvar et al., 2014; Moran et al., 2015; Onat et al., 2014b). These findings make it clear that the use of MRIO databases to model SCs for new or very specific products, like for instance carbon fiber reinforced plastic or biopolymers, would have different SCs than conventional plastics but due to the current sectoral aggregation in existing MRIO databases, no differentiation between those two would be possible. The indications of the case study as in this research should be put on a broader basis by carrying out a larger sample of case studies and matching them to the MRIO SC analyses. Besides expanding the data for the approach that was applied in this research, another route worth taking would be to define a representative product within an industry sector based on its share in transactions. That way real life case studies could be carried out for that product which could be compared more meaningfully with the MRIO analysis results for the SC of that respective industry sector.

In summary, the indication of the real case study showed that the theoretical modelling, be it in a more aggregated or disaggregated way by production tier, did only roughly represent the real SC. The suitability and robustness of MRIO based SC analyses for identifying SC structures was stronger when based on the CIC than when using the analysis by production tier as the broader focus also did not raise expectations for higher accuracy. This limits the possible applications of MRIO analysis for SC analysis, where detailed information on the structure of SC, i.e. the location of suppliers in regards to production tier, region and industry sector, are needed.

4.3.4. *Usefulness of SC analysis as basis for facilitating S-LCA*

The inception of this research was originally motivated by looking at MRIO databases for possible support in conducting S-LCA. Therefore, in this section, the results of this analysis are interpreted in the light of a possible application of MRIO analysis for facilitating S-LCA.

The goal of an S-LCA is to assess the social impacts of a product over its entire life cycle. To evaluate social impacts primary data are necessary, because social impacts are depending on the location of the analyzed production processes. As primary data collection for an entire SC would be an endeavor virtually impossible for complex products, a way has to be found to still gauge the social impacts of parts of the supply chain for which no primary data can be retrieved.

From the point of view of a practitioner, MRIO analysis would be an attractive way of limiting the efforts for primary data collection in S-LCA if (a) it delivers the definitive information of regional and industry sector distribution of suppliers and if (b) it does so even when only tier 1 suppliers are known. When the information on regional and industry sector distribution of suppliers are known, secondary data in the form of social risk databases can be used to identify social hotspots, i.e. life cycle stages and/or processes with increased risks for social violations. Examples for these databases are the Social Hotspots Database (SHDB) (Benoit-Norris et al., 2012; Benoît Norris, 2014; Benoít Norris et al., 2013), Product Social Impact Life Cycle Assessment (PSILCA) Database (Ciroth and Eisfeldt, 2015) and Maplecroft (Verisk Maplecroft, 2017). The social hotspots can be taken as indication on where to focus primary data collection for S-LCA and thus drastically reduce the data collection effort. This would mean a substantial facilitation of conducting S-LCA.

The fact that the results of the SC analyses did not capture the real life SC even remotely, renders the application of MRIO analysis useless for such a specific purpose.

Even though MRIO analysis has the potential to increase the understanding of otherwise unknown SC structures, they are "macroscopic" systems usually used for top-down analyses. Thus, the reliability of their application for "microscopic" or bottom-up analyses, e.g. on product level, should not be relied on too heavily (Moran et al., 2015; Stadler et al., 2014). This has been substantiated by this research for the practical application of MRIO databases for specific SC analysis. An additional drawback of MRIO analysis is the inability to differentiate between products of the same sector. For example, a practitioner will not be able to differentiate between SC for biopolymers vs. conventional plastics as they would fall into the same industry sector within MRIO databases (Kucukvar and Tatari, 2013; Lenzen et al., 2004; Onat et al., 2014a, 2014b; Reimann et al., 2010).

The findings – individual SCs are not correctly identified by MRIO analysis – might be no surprise to MRIO specialists. However, for other practitioners who are new to this field and look at a possible application of MRIO databases for detailed SC analysis this could be crucial.

The suitability of SC analysis to enable hotspot assessment could also be investigated using the working hour model (Benoit-Norris et al., 2012; Zimmer et al., 2017). It may be that SCs that are analyzed by production tier and based on working hours resemble a real world SC more closely than the analysis based on transaction flows as in this research.

4.4. *Limitations*

The results and conclusions presented here have to be interpreted in the context of the following limitations: The comparison of MRIO databases was based on a unified, aggregated form of the different MRIO databases (common classification system). As such the comparisons presented here are to some degree distorted by the manipulations to the original databases necessary to make them comparable.

Obviously, the comparison results based on SC analyses and the identified challenges only hold true for the specific premises of this research. It is possible that other ways of conducting SC analysis using MRIO would arrive at different results. Also the applied way of conduction the SC analysis by production tier in this research was no SPA in the classical sense but a restricted version as a limit for the number of suppliers per production tiers was set. Application of SPA could bring further insights into the way MRIO analysis of SCs should be conducted to gain meaningful information on SC.

The use of just one single case study to get an indication on the meaningfulness of the SC analyses is a clear limitation. Furthermore, the comparison of MRIO analysis results with a unique, real SC were more of indicative nature. A future research approach that identifies a representative product of an industry sector, i.e. the product that has the highest weight in that respective industry sector, and compares several real SCs supplying this product with the MRIO analysis results of that industry sector. It would be expected to yield more reliable results than this research did. As the case study selected was a SC with clearly associated and delineated sectors in MRIO databases, it was by no means chosen as a worst case example, but rather as the contrary. More complex case studies on SCs as just described should be conducted to expand the basis for interpretation. Also, comparisons with additional MRIO databases like OpenEU might underline or contradict the findings of this research.

5. Conclusions

The goal of this paper was to determine how MRIO databases differed in their structure and which way to approach SC analysis using MRIO databases would be preferable for practitioners. It was expected that through SC analysis meaningful information on SC structures could be gathered, i.e. the regional and sectoral distribution of otherwise unknown SC segments.

This research added scientific insights by conducting the first systematic comparison of transaction data contained in the four major MRIO databases (Eora, EXIOBASE, GTAP and WIOD). Additionally, no similar analysis of evaluating the consistency of SC analysis results between MRIO databases as carried out in this research is known to the authors. The findings brought forth several conclusions for researchers and practitioners in the field of MRIO analysis.

- The share of intra-regional transactions was consistent across databases but surprisingly high. Future research here would be of high interest as a clarification of this may lead to additional insights on either MRIO construction or the global connections of intermediate consumption.
- As the approach to constructing the MRIO appeared to be a deciding factor in the differences between MRIO databases, an analysis of the respective MRIO creation methods could enable a better understanding of the reasons and drivers that led to the differences in inter-regional transactions.
- MRIO analyses did not provide consistent results that were independent from a chosen database. Depending on the reason behind this, for practitioners this would either mean that they should carefully evaluate, whether the respective construction approach of the MRIO database could be beneficial or detrimental to their intended analysis. Or it would mean that the use of several MRIO databases is advised as an analysis result can only be regarded as robust if several MRIO analyses arrive at the same result. Future research could try to combine the strengths from MRIO databases and produce a consistent, disaggregated synthesis of all of them.
- Practitioners should rely on the more aggregated SC analysis based on CIC as the additional level of detail by SPA only gives the impression of additional insight, while it actually might lie further from the truth.
- The possible application of MRIO analysis to enable further SC analyses like S-LCA was found to be limited. For practitioners, this means that MRIO analysis should only be used for macroscopic evaluation and refrain from trying to gain insights into specific SCs.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jclepro.2018.06.082.

References

Aguiar, A., Narayanan, B., McDougall, R., 2016. An overview of the GTAP 9 data base. J. Glob. Econ. Anal. 1, 181–208. https://doi.org/10.21642/JGEA.010103AF.

Akkermans, H., Bogerd, P., van Doremalen, J., 2004. Travail, transparency and trust: a case study of computer-supported collaborative supply chain planning in high-tech electronics. Eur. J. Oper. Res. 153, 445–456. https://doi.org/10.1016/S0377-2217(03)00164-4.

Alsamawi, A., Murray, J., Lenzen, M., 2014. The employment footprints of nations: uncovering master-servant relationships. J. Ind. Ecol. 18, 59–70. https://doi.org/10.1111/jiec.12104.

Arto, I., Rueda-Cantuche, J.M., Peters, G.P., 2014. Comparing the GTAP-MRIO and WIOD databases for carbon footprint analysis. Econ. Syst. Res. 26, 327–353. https://doi.org/10.1080/09535314.2014.939949.

Benoit-Norris, C., Cavan, D.A., Norris, G.A., 2012. Identifying social impacts in product supply chains: overview and application of the social hotspot database. Sustainability 4, 1946–1965. https://doi.org/10.3390/su4091946.

Benoît Norris, C., 2014. Data for social LCA. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-013-0644-7.

Benoît Norris, C., Norris, G.A., Aulisio Cavan, D., 2013. Social Hotspots Database - Supporting Documentation.

Betts, T.K., Smith, L.M., Smith, K.T., 2018. Financial analysis of companies concerned about human rights. Int. J. Bus. Excel. 14, 360. https://doi.org/10.1504/IJBEX.2018.10010421.

BMW Group, 2015. Sustainable Value Report.

Bouwmeester, M.C., Oosterhaven, J., 2013. Specification and aggregation errors in environmentally extended input–output models. Environ. Resour. Econ. 56, 307–335. https://doi.org/10.1007/s10640-013-9649-8.

Ciroth, A., Eisfeldt, F., 2015. A New, Comprehensive Database for Social LCA. PSILCA.

Crama, Y., Defourny, J., Gazon, J., 1984. Structural decomposition of multipliers in input-output or social accounting matrix analysis. Econ. Appl. 37, 215–222.

Defourny, J., Thorbecke, E., 1984. Structural path analysis and multiplier decomposition within a social accounting matrix framework. Econ. J. 94 (111). https://doi.org/10.2307/2232220.

Dreyer, L., Hauschild, M., Schierbeck, J., 2006. A framework for social life cycle impact assessment. Int. J. Life Cycle Assess. 11, 88–97 (10 pp). https://doi.org/10.1065/lca2005.08.223.

Finkbeiner, M., Reimann, K., Ackermann, R., 2008. Life cycle sustainability assessment (LCSA) for products and processes. In: SETAC Europe 18th Annual Meeting, 25–29 May. Warsaw, Poland.

Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards life cycle sustainability assessment. Sustainability 2, 3309–3322. https://doi.org/10.3390/su2103309.

Gallego, B., Lenzen, M., 2009. Estimating generalized regional input–output systems: a case study of Australia. In: Ruth, M., Davidsdottir, B. (Eds.), The Dynamics of Regions and Networks in Industrial Ecosystems. Edward Elgar Publishing, Boston, MA, USA, pp. 55–82. https://doi.org/10.4337/9781848449374.00014.

Heijungs, R., Lenzen, M., 2014. Error propagation methods for LCA - a comparison. Int. J. Life Cycle Assess. 19, 1445–1461. https://doi.org/10.1007/s11367-014-0751-0.

Inomata, S., Owen, A., 2014. Comparative evaluation of MRIO databases. Econ. Syst. Res. 26, 239–244. https://doi.org/10.1080/09535314.2014.940856.

Jørgensen, A., Bocq, A. Le, Nazarkina, L., Hauschild, M., 2008. Methodologies for social life cycle assessment. Int. J. Life Cycle Assess. 13, 96–103. https://doi.org/10.1065/lca2007.11.367.

Jørgensen, A., Finkbeiner, M., Jørgensen, M.S., Hauschild, M.Z., 2010. Defining the baseline in social life cycle assessment. Int. J. Life Cycle Assess. 15, 376–384. https://doi.org/10.1007/s11367-010-0176-3.

Karlewski, H., 2016. Social Life Cycle Assessment in the Automotive Industry. TU Berlin. https://doi.org/. https://doi.org/10.14279/depositonce-4974.

Kitzes, J., Berlow, E., Conlisk, E., Erb, K., Iha, K., Martinez, N., Newman, E.A., Plutzar, C., Smith, A.B., Harte, J., 2017. Consumption-based conservation targeting: linking biodiversity loss to upstream demand through a global wildlife footprint. Conserv. Lett. 10, 531–538. https://doi.org/10.1111/con4.12321.

Klöpffer, W., 2008. Life cycle sustainability assessment of products. Int. J. Life Cycle Assess. 13, 89–95. https://doi.org/10.1065/lca2008.02.376.

Klöpffer, W., 2003. Life-Cycle based methods for sustainable product development. Int. J. Life Cycle Assess. 8, 157–159. https://doi.org/10.1007/BF02978462.

Klöpffer, W., Grahl, B., 2014. From LCA to sustainability assessment. In: Life Cycle Assessment (LCA). Wiley-VCH Verlag GmbH & Co. KGaA, Weinheim, Germany, pp. 357–374. https://doi.org/10.1002/9783527655625.ch6.

Koplin, J., Seuring, S., Mesterharm, M., 2007. Incorporating sustainability into supply management in the automotive industry – the case of the Volkswagen AG. J. Clean. Prod. 15, 1053–1062. https://doi.org/. https://doi.org/10.1016/j.jclepro.2006.05.024.

Kucukvar, M., Noori, M., Egilmez, G., Tatari, O., 2014. Stochastic decision modeling for sustainable pavement designs. Int. J. Life Cycle Assess. 19, 1185–1199. https://doi.org/10.1007/s11367-014-0723-4.

Kucukvar, M., Tatari, O., 2013. Towards a triple bottom-line sustainability assessment of the U.S. construction industry. Int. J. Life Cycle Assess. 18, 958–972. https://doi.org/10.1007/s11367-013-0545-9.

Lamming, R.C., Caldwell, N.D., Harrison, D.A., Phillips, W., 2001. Transparency in supply relationships: concept and practice. J. Supply Chain Manag. 37, 4–10. https://doi.org/10.1111/j.1745-493X.2001.tb00107.x.

Lenzen, M., 2011. Aggregation versus disaggregation in input-output analysis of the environment. Econ. Syst. Res. 23, 73–89. https://doi.org/10.1080/09535314.2010.548793.

Lenzen, M., 2002. A guide for compiling inventories in hybrid life-cycle assessments: some Australian results. J. Clean. Prod. 10, 545–572. https://doi.org/10.1016/S0959-6526(02)00007-0.

Lenzen, M., Crawford, R., 2009. The path exchange method for hybrid LCA. Environ. Sci. Technol. 43, 8251–8256. https://doi.org/10.1021/es902090z.

Lenzen, M., Dey, C., 2000. Truncation error in embodied energy analyses of basic iron and steel products. Energy 25, 577–585. https://doi.org/10.1016/S0360-5442(99)00088-2.

Lenzen, M., Kanemoto, K., Moran, D., Geschke, A., 2012. Mapping the structure of the world economy. Environ. Sci. Technol. 46, 8374–8381. https://doi.org/10.1021/es300171x.

Lenzen, M., Moran, D., Kanemoto, K., 2013. Building Eora: a global multi-regional input-output database at high country and sector resolution. Econ. Syst. Res. 25, 20–49. https://doi.org/10.1080/09535314.2013.769938.

Lenzen, M., Pade, L.-L., Munksgaard, J., 2004. CO2 multipliers in multi-region input-output models. Econ. Syst. Res. 16, 391–412.

Lenzen, M., Treloar, G., 2003. Differential convergence of life-cycle inventories toward upstream production layers. J. Ind. Ecol. 6, 137–160. https://doi.org/10.1162/108819802766269575.

Lenzen, M., Treloar, G., 2002. Embodied energy in buildings: wood versus concrete—reply to Börjesson and Gustavsson. Energy Pol. 30, 249–255. https://doi.org/10.1016/S0301-4215(01)00142-2.

Leontief, W., Strout, A., 1963. Multiregional input-output analysis. In: Barna, T. (Ed.), Structural Interdependence and Economic Development: Proceedings of an International Conference on Input-output Techniques, Geneva, September 1961. Palgrave Macmillan, UK, London, pp. 119–150. https://doi.org/10.1007/978-1-349-81634-7_8.

Martínez-Blanco, J., Lehmann, A., Chang, Y.-J., Finkbeiner, M., 2015. Social organizational LCA (SOLCA)—a new approach for implementing social LCA. Int. J. Life Cycle Assess. 20, 1586–1599. https://doi.org/10.1007/s11367-015-0960-1.

Miller, R.E., Shao, G., 1990. Spatial and sectoral aggregation in the commodity - industry multiregional input - output model. Environ. Plan. A 22, 1637–1656. https://doi.org/10.1068/a221637.

Moran, D., McBain, D., Kanemoto, K., Lenzen, M., Geschke, A., 2015. Global supply chains of coltan. J. Ind. Ecol. 19, 357–365. https://doi.org/10.1111/jiec.12206.

Moran, D., Wood, R., 2014. Convergence between the EORA, WIOD, EXIOBASE, and OPENEU's consumption-based carbon accounts. Econ. Syst. Res. 26, 245–261. https://doi.org/10.1080/09535314.2014.935298.

Narayanan, B.G., Aguiar, A., McDougall, R., 2012. Global Trade, Assistance, and Production: the GTAP 8 Data Base. Purdue University.

Onat, N., Kucukvar, M., Tatari, O., 2014a. Integrating triple bottom line input–output analysis into life cycle sustainability assessment framework: the case for US buildings. Int. J. Life Cycle Assess. 19, 1488–1505. https://doi.org/10.1007/s11367-014-0753-y.

Onat, N., Kucukvar, M., Tatari, O., 2014b. Towards life cycle sustainability assessment of alternative passenger vehicles. Sustainability 6, 9305–9342. https://doi.org/10.3390/su6129305.

Owen, A., Steen-Olsen, K., Barrett, J., Wiedmann, T., Lenzen, M., 2014. A structural decomposition approach to comparing MRIO databases. Econ. Syst. Res. 26, 262–283. https://doi.org/10.1080/09535314.2014.935299.

Owen, A., Wood, R., Barrett, J., Evans, A., 2016. Explaining value chain differences in MRIO databases through structural path decomposition. Econ. Syst. Res. 28, 243–272. https://doi.org/10.1080/09535314.2015.1135309.

Parmigiani, A., Klassen, R.D., Russo, M.V., 2011. Efficiency meets accountability: performance implications of supply chain configuration, control, and capabilities. J. Oper. Manag. 29, 212–223. https://doi.org/10.1016/j.jom.2011.01.001.

Peters, G.P., Andrew, R., Lennox, J., 2011. Constructing an environmentally extended multi-regional input–output table using the GTAP database. Econ. Syst. Res. 23, 131–152. https://doi.org/10.1080/09535314.2011.563234.

Reimann, K., Finkbeiner, M., Horvath, A., Matsuno, Y., 2010. Evaluation of Environmental Life Cycle Approaches for Policy and Decision Making Support in Micro and Macro Level Applications. https://doi.org/10.2788/32275.

Simas, M., Golsteijn, L., Huijbregts, M., Wood, R., Hertwich, E., 2014. The "bad labor" footprint: quantifying the social impacts of globalization. Sustainability 6, 7514–7540. https://doi.org/10.3390/su6117514.

Stadler, K., Steen-Olsen, K., Wood, R., 2014. The "Rest of the World" - estimating the economic structure of missing regions in global multi-rgional input-output tables. Econ. Syst. Res. 26, 303–326. https://doi.org/10.1080/09535314.2014.936831.

Stadler, K., Wood, R., Bulavskaya, T., Södersten, C.-J., Simas, M., Schmidt, S., Usubiaga, A., Acosta-Fernández, J., Kuenen, J., Bruckner, M., Giljum, S., Lutter, S., Merciai, S., Schmidt, J.H., Theurl, M.C., Plutzar, C., Kastner, T., Eisenmenger, N., Erb, K.-H., de Koning, A., Tukker, A., 2018. EXIOBASE 3: developing a time series of detailed environmentally extended multi-regional input-output tables. J. Ind. Ecol. https://doi.org/10.1111/jiec.12715.

Steen-Olsen, K., Owen, A., Barrett, J., Guan, D., Hertwich, E.G., Lenzen, M., Wiedmann, T., 2016. Accounting for value added embodied in trade and consumption: an intercomparison of global multiregional input–output databases. Econ. Syst. Res. 28, 78–94. https://doi.org/10.1080/09535314.2016.1141751.

Steen-Olsen, K., Owen, A., Hertwich, E.G., Lenzen, M., 2014. Effects of sector aggregation on CO2 multipliers in multiregional input-output analyses. Econ. Syst. Res. 26, 284–302. https://doi.org/10.1080/09535314.2014.934325.

Su, B., Huang, H.C., Ang, B.W., Zhou, P., 2010. Input–output analysis of CO2 emissions embodied in trade: the effects of sector aggregation. Energy Econ. 32, 166–175. https://doi.org/10.1016/j.eneco.2009.07.010.

Suh, S., Heijungs, R., 2007. Power series expansion and structural analysis for life cycle assessment. Int. J. Life Cycle Assess. 12, 381–390. https://doi.org/10.1065/lca2007.08.360.

Suh, S., Lenzen, M., Treloar, G.J., Hondo, H., Horvath, A., Huppes, G., Jolliet, O., Klann, U., Krewitt, W., Moriguchi, Y., Munksgaard, J., Norris, G., 2004. System boundary selection in life-cycle inventories using hybrid approaches. Environ. Sci. Technol. 38, 657–664. https://doi.org/10.1021/es0263745.

Systain, 2018. Estell - Social Footprinting [WWW Document]. URL. https://en.systain.com/estell-social-footprinting/.

The World Bank, 2017. Trade (% of GDP) [WWW Document]. http://data.worldbank.org/indicator/NE.TRD.GNFS.ZS (Accessed 26 January 2017).

Timmer, M.P., Dietzenbacher, E., Los, B., Stehrer, R., de Vries, G.J., 2015. An illustrated user guide to the world input-output database: the case of global automotive production. Rev. Int. Econ 23, 575–605. https://doi.org/10.1111/roie.12178.

Tukker, A., Dietzenbacher, E., 2013. Global multiregional input-output frameworks: an introduction and outlook. Econ. Syst. Res. 25, 1–19. https://doi.org/10.1080/09535314.2012.761179.

UNEP/SETAC Life Cycle Initiative, 2011. Towards a Life Cycle Sustainability Assessment - Making Informed Choices on Products. UNEP/SETAC Life Cycle Initiative, France.

Verisk Maplecroft, 2017. Verisk Maplecroft [WWW Document]. https://maplecroft.com/.

Waugh, F.V., 1950. Inversion of the Leontief matrix by power series. Econometrica 18, 142. https://doi.org/10.2307/1907265.

Wiebe, K.S., Lenzen, M., 2016. To RAS or not to RAS? What is the difference in outcomes in multi-regional input–output models? Econ. Syst. Res. 28, 383–402. https://doi.org/10.1080/09535314.2016.1192528.

Wiedmann, T., Wilting, H.C., Lenzen, M., Lutter, S., Palm, V., 2011. Quo Vadis MRIO? Methodological, data and institutional requirements for multi-region input-output analysis. Ecol. Econ. 70, 1937–1945. https://doi.org/10.1016/j.ecolecon.2011.06.014.

Wolf, J., 2014. The relationship between sustainable supply chain management, stakeholder pressure and corporate sustainability performance. J. Bus. Ethics 119, 317–328. https://doi.org/10.1007/s10551-012-1603-0.

Wood, R., Lenzen, M., 2009. Structural path decomposition. Energy Econ. 31, 335–341. https://doi.org/10.1016/j.eneco.2008.11.003.

Wood, R., Stadler, K., Bulavskaya, T., Lutter, S., Giljum, S., de Koning, A., Kuenen, J., Schütz, H., Acosta-Fernández, J., Usubiaga, A., Simas, M., Ivanova, O., Weinzettel, J., Schmidt, J., Merciai, S., Tukker, A., 2014. Global sustainability accounting—developing EXIOBASE for multi-regional footprint analysis. Sustainability 7, 138–163. https://doi.org/10.3390/su7010138.

Wood, R., Stadler, K., Simas, M., Bulavskaya, T., Giljum, S., Lutter, S., Tukker, A., 2018. Growth in environmental footprints and environmental impacts embodied in trade: resource efficiency indicators from EXIOBASE3. J. Ind. Ecol. https://doi.org/10.1111/jiec.12735.

Wyckoff, A., Roop, J., 1994. The embodiment of carbon in imports of manufactured products: implications for international agreements on greenhouse gas emissions. Energy Pol. 22, 187–194.

Zimmer, K., Fröhling, M., Breun, P., Schultmann, F., 2017. Assessing social risks of global supply chains: a quantitative analytical approach and its application to supplier selection in the German automotive industry. J. Clean. Prod. 149, 96–109. https://doi.org/10.1016/j.jclepro.2017.02.041.

Zimmer, K., Fröhling, M., Schultmann, F., 2016. Sustainable supplier management – a review of models supporting sustainable supplier selection, monitoring and development. Int. J. Prod. Res. 54, 1412–1442. https://doi.org/10.1080/00207543.2015.1079340.

2.1.3 Alternative Approach to Social Hotspot Assessment

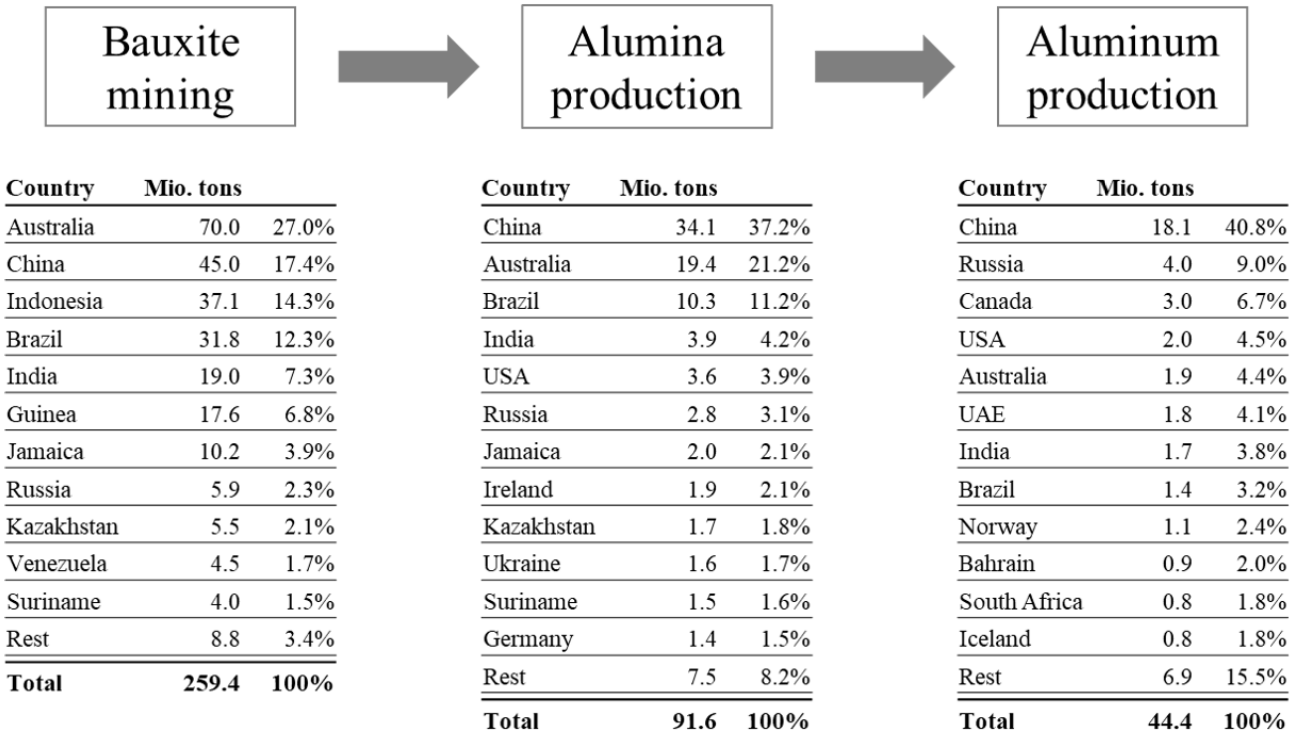
As MRIO analysis was found to offer no substantial support for social hotspot assessment (c.f. section 2.1.2), an alternative approach was needed to be able to assess the social risks associated with a product or component. Therefore, an approach was developed within this dissertation that was based on materials and their origins in the world. The approach relied on the social topics and indicators as identified in section 2.1.1 and is presented in the following.

First, data sources were evaluated (section 2.1.3.1), followed by a quantification of indicator results (section 2.1.3.2). The centerpiece of the approach is presented in section 2.1.3.3 which shows the calculation logic of how to obtain a combined risk score per material from the selected indicators and data sources. It is also shown how this approach can be used to get an indication about the social impacts of vehicle components (section 2.1.3.4). The results of the approach are presented in section 2.1.3.5.

2.1.3.1 Data Sources

For the evaluation of social risks, the social risk assessment data for countries and industry sectors from the SHDB was taken. To adequately appraise the social risk associated with materials used by and automotive company (e.g. steel, aluminum, lithium, copper, plastics) all relevant materials for the production of a vehicle were identified. To be able to assess what risk is associated with buying those materials from the worldwide market, the risk-data from the SHDB was combined with the material production data, i.e. the information on material volume and the respective country it originated from. Production data was gathered so that 80% of the world production was accounted for. Figure 6 shows the exemplary production chain of aluminum, consisting of bauxite mining, alumina production and aluminum production.

For the bauxite mining step, for example, eleven countries accounted for 96.6% of the world's production volume. As the SHDB distinguishes industry sectors, the respective production steps were matched to the available industry sectors in the SHDB. For example, for the step of bauxite mining of the aluminum production social risks for the mining sector were used. The step of aluminum production, however, was characterized with social risk data for the metal sector. The indicators covered were already introduced in section 2.1.1.



Bauxite mining

Country	Mio. tons	
Australia	70.0	27.0%
China	45.0	17.4%
Indonesia	37.1	14.3%
Brazil	31.8	12.3%
India	19.0	7.3%
Guinea	17.6	6.8%
Jamaica	10.2	3.9%
Russia	5.9	2.3%
Kazakhstan	5.5	2.1%
Venezuela	4.5	1.7%
Suriname	4.0	1.5%
Rest	8.8	3.4%
Total	**259.4**	**100%**

Alumina production

Country	Mio. tons	
China	34.1	37.2%
Australia	19.4	21.2%
Brazil	10.3	11.2%
India	3.9	4.2%
USA	3.6	3.9%
Russia	2.8	3.1%
Jamaica	2.0	2.1%
Ireland	1.9	2.1%
Kazakhstan	1.7	1.8%
Ukraine	1.6	1.7%
Suriname	1.5	1.6%
Germany	1.4	1.5%
Rest	7.5	8.2%
Total	**91.6**	**100%**

Aluminum production

Country	Mio. tons	
China	18.1	40.8%
Russia	4.0	9.0%
Canada	3.0	6.7%
USA	2.0	4.5%
Australia	1.9	4.4%
UAE	1.8	4.1%
India	1.7	3.8%
Brazil	1.4	3.2%
Norway	1.1	2.4%
Bahrain	0.9	2.0%
South Africa	0.8	1.8%
Iceland	0.8	1.8%
Rest	6.9	15.5%
Total	**44.4**	**100%**

Figure 6: Production data for material-based assessment. Example of aluminum: production data is broken down by production step and main producing countries (United States Geological Survey, 2013).

2.1.3.2 Quantification of SHDB Data

As the risk levels within the SHDB were of semi-quantitative nature, they were transformed to quantitative metrics to enable further calculations. Table 3 shows the quantification logic to assign values to the SHDB classifications. The scale to which the semi-quantitative indicators were transformed was taken from the "Handbook for Product Social Impact Assessment" (Fontes et al., 2015).

Table 3: Logic to transform SHDB risk classification into quantitative scores

SHDB Classification	Score
"No Evidence"	+2
"Low"	+1
"Medium"	0
"High"	-1
"Very High"	-2

2.1.3.3 Composite Material Risk Score

The quantified scores from the SHDB were combined to form a social risk score for the 4 categories "Human Rights", "Social Standards", "Health and Safety" and "Corruption" (see section 2.1.1) per country i, production step j and material k. The combination logic to calculate a score per category is laid out in the following sections. The following indicators constituted the social topic "Human Rights":

- Risk of child labor$_{i,j,k}$ = "Child"
- Risk of forced labor$_{i,j,k}$ = "Forced"
- Risk of violation of right to collective bargaining$_{i,j,k}$ = "Bargaining"
- Risk of violation of right of freedom of association$_{i,j,k}$ = "Association"

They were gathered for country i, production step j and material k, respectively. After the transformation into quantitative metrics (c.f. section 2.1.3.2), the four indicators were aggregated into one risk score for the "Human Rights" topic (c.f. equation 5).

$$\begin{aligned}\text{RS-HR}_{i,j,k} = IF \quad & \text{''}Child\text{''} < -1 \quad OR \quad \text{''}Forced\text{''} < -1; \\ & THEN -2; \\ & ELSE \quad (\text{''}Child\text{''} + \text{''}Forced\text{''} + \text{''}Bargaining\text{''} + \text{''}Association\text{''})/4;\end{aligned} \tag{5}$$

RS-HR$_{i,j,k}$ = risk score "Human Rights" for country i, production step j and material k

The risk of child or forced labor was deemed a knockout criterion (KO), meaning that if one of those indicators constituted a very high risk, the risk score for the entire social topic would also be set to a very high risk, i.e. a score of -2. Otherwise, the average of all four indicators was taken. For the topic "Social Standards" the risk score (RS-SO$_{i,j,k}$) for country i, production step j and material k was calculated by taking the average score of the two indicators "Potential of Average wage being < Minimum Wage" and "Risk of excessive working time".

For the topic "Health & Safety" the risk score (RS-HS$_{i,j,k}$) for country i, production step j and material k was calculated by taking the average score of the two indicators "Risk of non-fatal injuries" and "Risk of fatal injuries". For the risk score "Corruption" for country i, production step j and material k (RS-CO$_{i,j,k}$) the value of the indicator "Overall Risk for Corruption" was taken. In a next step, the four risk scores were aggregated into one score using the following logic:

$$\begin{aligned}\text{RS}_{i,j,k} = IF \quad & \text{''RS-HR}_{i,j,k}\text{''} < -1 \\ & THEN -2; \\ & ELSE \quad (\text{''RS-HR}_{i,j,k}\text{''} + \text{''RS-SO}_{i,j,k}\text{''} + \text{''RS-HS}_{i,j,k}\text{''} + \text{''RS-CO}_{i,j,k}\text{''})/4;\end{aligned} \tag{6}$$

RS$_{i,j,k}$ = Aggregated risk score for country i, production step j and material k

That means that the KO criterion of possible child or forced labor was also carried to the aggregated level. The combined social risk for a production step j of a material k (RS$_{j,k}$) was determined using the following formula:

$$\text{RS}_{j,k} = \frac{\sum_{i=1}^{n}(x_{i,j,k} * RS_{i,j,k})}{\sum_{i=1}^{n}(x_{i,j,k})} \tag{7}$$

n = number of countries in production step j of material k

x$_{i,j,k}$ = production volume in country i and production step j of material k

The composite material risk score for the production chain of a material k (RS_k) was determined by taking the average of all production steps:

$$\mathrm{RS}_k = \frac{\sum_{j=1}^{m}(x_{j,k} * RS_{j,k})}{\sum_{j=1}^{m}(x_{j,k})} \tag{8}$$

m = number of production steps of material k
$x_{j,k}$ = total production volume in production step j of material k

This material risk score can range from -2 ("very high risk") to +2 ("no evidence of risk"). By using this score, the potentially critical conditions in the production chain of a material (= hotspots) were identified to guide efforts to collect primary data for these hotspots.

2.1.3.4 Application of Material-Based Social Hotspot Assessment to Determine Social Risk of Components

The results of the material-related social risk scores were used to calculate an overall social risk score for every component of a vehicle. This was done to get an indication on the potential social impact caused by production of the components. As a starting point, the material composition of every component was determined. Then, the amount of every material in a component was multiplied by the respective material risk score and summed up to an overall social risk score of that component. Table 4 shows the calculation logic for the overall social risk score at the component level.

Table 4: Calculation logic to determine an overall social risk score at the component level

Component	**Quantity Material 1**	**Risk Score Material 1**	**Quantity Material 2**	**Risk Score Material 2**	**...**	**Overall Social Risk Score**
A	a kg	b	c kg	d	...	**a*b + c*d + ...**
B	e kg	b	f kg	d	...	**e*b + f*d + ...**
C	g kg	b	h kg	d	...	**g*b + h*d + ...**
...	...	...	...	...	...	**...**

2.1.3.5 Risk Scores of Metals and Components

Table 5 shows the results for the risk scores (RS_k) determined for metals relevant for vehicle production.

Table 5: Material risk scores of metals. The highest possible score was +2, the lowest possible score -2.

Material	**Risk Score ($\mathbf{RS}_k$)**
Aluminum	-0.72
Cobalt	-1.34
Copper	-0.67
Gold	-0.89
Palladium	-1.84
Platinum	-2.00
Rhodium	-2.00
Steel	-0.94
Tantalum	-1.53
Tin	-1.41
Tungsten	-0.50

Platinum and Rhodium showed the highest social risk scores as their world production was not as widespread — more than 80% of the world's Platinum production, for instance, was located in the two countries South Africa and the Russian Federation — and exhibited the risk of human rights violation — South Africa was assigned a very high risk for child labour by the SHDB, the Russian Federation was given a very high risk for forced labour and corruption. The next critical metals were Palladium, Copper, Tantalum and Tin. Interestingly, Tungsten — classified as conflict mineral (OECD, 2012) — was not as critical in this analysis as the critical country (Democratic Republic of the Congo) only accounted for <1 percent of the worlds production volume in 2012 (United States Geological Survey, 2013). Table 6 shows the exemplary results of the overall social risk scores at the component level.

Table 6: Example for the application of the social risk assessment approach. The table shows fictional values.

Component	**Quantity Aluminum**	**Risk Score Aluminum**	**Quantity Copper**	**Risk Score Copper**	**...**	**Overall Social Risk Score**
A	5 kg	-0.72	2 kg	-0.67	...	**-4.91**
B	3 kg	-0.72	1 kg	-0.67	...	**-2.83**
C	0 kg	-0.72	0 kg	-0.67	...	**0.0**
...	...	...	...	...	...	**...**

As can be seen in Table 6 the material-based social risk assessment approach enabled a quantitative evaluation of social risk related to the material composition of the individual components. The overall social risk score for each component from Table 6 can be evaluated relative to all other components in the vehicle using VIKOR (following the conceptual approach introduced in section 2) to obtain the criticality scores for the S-LCA dimension for every component. The order of assessment steps for the S-LCA impacts of components is summarized as follows:

1. Identification of material-related social risk score (RS)
2. Calculation of overall social risk score per component based on material composition of that component
3. Translation of the overall social risk score to S-LCA criticality points per component

How the results of this section feed into the improvement of the applicability of the LCSA framework is shown in section 2.4. In this section, an approach to conduct social hotspot assessment based on the worldwide production data of materials relevant to the automotive industry was introduced. It was furthermore shown, how the social risks that were determined by this approach can be applied to determine an overall social risk score per component that can be translated into the S-LCA criticality points for the LCSA framework. That means that the assessment of the impacts on the S-LCA dimension in the operationalized LCSA framework at component level is possible. However the support for decision makers in interpreting the LCSA results has still to be addressed. How this outstanding gap was tackled is shown in the next section.

2.2 Method to Interpret LCSA Results for Decision Makers

This section deals with the second research objective (c.f. 1.5) and presents the results of a publication by Tarne et al. (2018b) on the weighing of sustainability dimensions, i.e. results of LCA, LCC and S-LCA. The publication investigated how sustainability dimensions are considered in product-related decision making within an automotive company. The authors approached this research question by asking 54 decision makers about how they made decisions when trade-offs occurred between the three LCSA dimensions. Limit Conjoint Analysis was applied, asking respondents to rank different alternatives of a component with different impacts on the three sustainability dimensions according to their preference of use within the vehicle. From the ranking of the alternatives, the weighting of the respective dimensions could be derived. This was done by comparing the average rank of an individual attribute specification (e.g. lower price) with the rank that would be expected if this attribute specification was treated equally to the others. The results showed that no clear consensus across respondents could be elicited as the individual weightings were scattered. The average weighting of the sample came close to an equal weighting of all three sustainability dimensions with 34% for the economic, 35% for the environmental and 31% for the social dimension.

The paper addressed the research objective 2 (c.f. section 1.5) in the following way:

- **Determine the weights decision makers assign to the individual sustainability dimensions**
  A Limit Conjoint Analysis was carried out, asking 54 decision makers at an automotive company how they would weight sustainability dimensions in product related decisions.

- **Choose a weighting set to adopt in the LCSA framework**
  From the individual weightings of decision makers, a weighting set was identified that represented the quantification of the decision making process from different departments within the company. That set was chosen to be adopted in the LCSA framework as the acceptance of this weighting set within the respective company was expected to be high.

The publication by Tarne et al. (2018b) is presented on pages 71–84.

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In this section, a weighting set was chosen that enables the aggregation of the criticality points in the individual dimensions of the LCSA framework (LCA, LCC and S-LCA) into an overall LCSA criticality score. How this overall LCSA criticality score can support decision makers in interpreting the LCSA results is presented in section 2.4. The next section shows how the last missing piece — the integration of the LCSA results into product-related decision processes at an automotive company was addressed.

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LIFE CYCLE SUSTAINABILITY ASSESSMENT

Introducing weights to life cycle sustainability assessment—how do decision-makers weight sustainability dimensions?

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Abstract

Purpose Decisions based on life cycle sustainability assessment (LCSA) pose a multi-criteria decision issue, as impacts on the three different sustainability dimensions have to be considered which themselves are often measured through several indicators. To support decision-making at companies, a method to interpret multi-criteria assessment and emerging trade-offs would be beneficial. This research aims at enabling decision-making within LCSA by introducing weights to the sustainability dimensions.
Methods To derive weights, 54 decision-makers of different functions at a German automotive company were asked via limit conjoint analysis how they ranked the economic, environmental, and social performance of a vehicle component. Results were evaluated for the entire sample and by functional clusters. Additionally, sustainability respondents, i.e., respondents that dealt with sustainability in their daily business, were contrasted with non-sustainability respondents. As a last step, the impact of outliers was determined. From this analysis, practical implications for ensuring company-optimal decision-making in regard to product sustainability were derived.
Results and discussion The results showed a large spread in weighting without clear clustering. On average, all sustainability dimensions were considered almost equally important: the economic dimension tallied at 33.5%, the environmental at 35.2%, and the social at 31.2%. Results were robust as adjusting for outliers changed weights on average by less than 10%. Results by function showed low consistency within clusters hinting that weighting was more of a personal than a functional issue. Sustainability respondents weighted the social before the environmental and economic dimension while non-sustainability respondents put the economic before the other two dimensions. Provided that the results of this research could be generalized, the retrieved weighting set was seen as a good way to introduce weights into an operationalized LCSA framework as it represented the quantification of the already existing decision process. Therefore, the acceptance of this weighting set within the respective company was expected to be increased.
Conclusions It could be shown that conjoint analysis enabled decision-making within LCSA by introducing weights to solve a multi-criteria decision issue. Furthermore, implications for practitioners could be derived to ensure company-optimal decision-making related to product sustainability. Future research should look at expanding the sample size and geographical scope as well as investigating the weighting of indicators within sustainability dimensions and the drivers that influence personal decision-making in regard to weighting sustainability dimensions.

Keywords Automotive company · Conjoint analysis · LCSA · Limit conjoint analysis · MCDA · Multi-criteria decision analysis · Sustainability dimensions · Weighting

1 Introduction

Life cycle sustainability assessment (LCSA) is a framework that enables product sustainability assessment. It assesses the impacts of a product on the three dimensions of sustainability, i.e., economy, environment, and society, over its entire life cycle (Klöpffer 2003; Finkbeiner et al. 2008; Klöpffer 2008; Finkbeiner et al. 2010; UNEP/SETAC Life Cycle Initiative

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2011). In order to successfully support decision-makers in choosing the most sustainable product alternative, a clear communication of LCSA results is required (Finkbeiner et al. 2010; Traverso et al. 2012; Klöpffer and Grahl 2014; Traverso et al. 2015). Within LCSA trade-offs can occur, for instance, when one product alternative performs better in terms of environmental impacts while another one performs better regarding social impacts (Heijungs et al. 2010). Trade-offs can occur between the three dimensions of sustainability when assessing a product as well as among indicators for the same sustainability dimension (Ostermeyer et al. 2013). As LCSA delivers results for all three sustainability dimensions, which themselves can consist of several indicators, a decision based on LCSA poses a multi-criteria decision issue or at least a multi-criteria interpretation issue. Therefore, to better support decision-makers, a clear method to solve multi-criteria decision-making and identifying the preferable alternative in case of trade-offs would be beneficial (Bond et al. 2012; Tarne et al. 2017). One approach towards solving multi-criteria decision issues is introducing weights to the criteria in order to be able to come to an aggregated evaluation (Yang et al. 2017). As the introduction of weights enables aggregated evaluation of impacts on all three sustainability dimensions, the concept of weak sustainability is assumed. The difference between the strong and weak interpretation of sustainability is that in the weak interpretation, improvements of impacts on one dimension can compensate for deterioration of impacts on other dimensions. The strong interpretation on the other hand assumes a hierarchical relationship between sustainability dimensions allowing for no compensation (Singh et al. 2009; JRC 2012). In order to implement an operationalized LCSA framework at an automotive company, the identification of the company-optimal weighting of sustainability criteria would support the decision-making process.

The goal of this paper is to enable decision-making within an operationalized LCSA framework through determining the weights of sustainability dimensions at an automotive company by questioning decision-makers on how they evaluate trade-offs between these dimensions. The focus laid on finding the weighting of the three sustainability dimensions against each other and not the weighting of individual indicators within dimensions.

2 Background

There are many methods available for eliciting weights (Alfares and Duffuaa 2008), like the data envelopment analysis (Yang et al. 2017), the critic method (Diakoulaki et al. 1995), Simon's procedure (Figueira and Roy 2002), and, most notably, the analytical hierarchy process (Choo et al. 1999; Scholl et al. 2005; Alfares and Duffuaa 2008) as well as conjoint analysis (Scholl et al. 2005; Alfares and Duffuaa 2008). Conjoint analysis is usually used when trying to determine the impact of selected product attributes on the preference of consumers (Cattin and Wittink 1982; Wittink and Cattin 1989). As conjoint analysis is designed to determine the weight that every product feature has in the purchasing decision of the customer (Skiera and Gensler 2002; Baier and Brusch 2009; Backhaus et al. 2011b) and most realistically resembles the decision process for a product (Baier and Brusch 2009), this method was also chosen for this research.

Conjoint analysis has been used to evaluate ecosystem services since the 1990s (Alriksson and Öberg 2008). Itsubo et al. (2004) were the first to apply conjoint analysis in connection with life cycle impact assessment (LCIA) in the course of the development of the LIME method (Itsubo and Inaba 2003) to determine weighting factors for environmental impacts. They aimed at determining how the Japanese population valued different safeguard subjects. Therefore, they questioned 400 Japanese citizens via discrete choice experiments to determine the utility of four endpoint categories (safeguard subjects) in environmental LCIA. They conducted a larger, more representative study for the update to the LIME2 method (Itsubo et al. 2012). The goal and scope of the study at hand deviates from previous studies (Itsubo et al. 2004; Mettier and Hofstetter 2004; Mettier et al. 2006; Mettier and Scholz 2008; Itsubo et al. 2012) which focused on the evaluation of environmental impacts, while this study aims at evaluating the three dimensions of sustainability. Furthermore, the respondents were decision-makers at an automotive company, meaning that the results of this study offer insights in the relevance of the sustainability dimensions in industry decision-making. "Conjoint analysis" stands for CONsidered JOINTly, meaning that test subjects are not asked explicitly about the evaluation of an individual product attribute but that several attributes are presented and evaluated together. A product can have different attributes like price, color, or taste. These attributes then can have different specifications (attribute specification), e.g., blue, red, or green for the attribute color.

The choice of product attributes and their specifications are crucial for the results of the analysis. Backhaus et al. (2011a) list several criteria that should be considered:

- Relevance: attributes should have relevance for purchase decision
- Controllability: the manufacturer should be able to influence the attribute
- Independence from preference: utility of an attribute should be independent from other attributes
- Realistic: attributes should be realizable
- Compensatory: one attribute can compensate for another
- No knockout (KO) criterion: an attribute or a specification of it should not lead to immediate rejection of the product

KO criteria should be avoided within conjoint analysis as they over-proportionally influence the stated preference and thus the results (Kohne et al. 2005; Baier and Brusch 2009; Backhaus et al. 2011a).

With conjoint analysis, the overall preference for an entire product is evaluated. As this is done for differently configured products, the utility of the individual attribute can be derived through decomposing the evaluated product alternatives (Green and Srinivasan 1978). This decompositional approach is regarded as closer to the real purchasing decision than compositional approaches, i.e. approaches that directly inquire about product attributes and specifications (Baier and Brusch 2009). The Conjoint Analysis therefore asks respondents to evaluate different combinations of product attributes and their specifications. These combinations are called stimuli. The traditional conjoint analysis retrieves the preferences of stimuli through stated preference by respondents, e.g., their ranking or rating of several stimuli (Backhaus et al. 2011b). The number of these stimuli should not be too excessive so as not to overwhelm respondents in their evaluation (Johnson and Orme 1996). This can be achieved by applying a so-called reduced or factorial design that is developed out of the initial amount of stimuli (Adepoju and Ipinyomi 2016). The aim is to find a set of stimuli that is representative for the entire number of possible stimuli.

3 Methods

3.1 Limit conjoint analysis

The limit conjoint analysis works by asking respondents to rank different product alternatives, represented by stimulus cards, by the order in which they would consider buying them. Additionally, they are asked to place a "limit card" that separates the stimuli that they would actually consider buying from those that they would not consider buying. By relating the order of preferences to the variation in product attributes, the importance or weight of each attribute for the purchasing decision can be determined (Sichtmann and Stingel 2007). The utility values for the individual attribute specifications are calculated from the different total utility values of the presented stimuli (decompositional approach).

After the ranking of the stimulus cards, the stimuli are assigned a total utility value u_j based on their relative position to the limit card. The utility values are assigned in a way that with every ranking position, the utility value changes by one (Voeth and Hahn 1998; Voeth 2000). As the limit card indicates the threshold between purchasing and not purchasing, the stimulus above it is assigned the value 0.5 and the value below it − 0.5. The limit card itself is not counted in the ranking. The following formula shows the logic of the utility value assignment:

$$u_j = R_L - 0.5 - R_j \tag{1}$$

u_j = total utility value of stimulus j R_L = rank before which the limit card was set R_j = rank of stimulus j

The utility values of individual attribute specifications can be estimated by the following steps that are based on Baier and Brusch (2009). First, the average utility value over all stimuli (μ) is determined:

$$\mu = \frac{\sum_{j=1}^{J} u_j}{J} \tag{2}$$

μ = average utility value over all stimuli J = number of presented stimuli

Second, the average utility value is taken from all stimuli where the respective attribute specification is present.

$$\varphi_{k,l} = \frac{\sum_{j=1}^{J} u_j \cdot v_{j,k,l}}{\sum_{j=1}^{J} v_{j,k,l}} \tag{3}$$

$\varphi_{k,l}$ = average utility of all stimuli that contain attribute k with specification l

$$v_{j,k,l} = \begin{cases} 1 \text{ if attribute k with specification l was present in stimulus j} \\ 0 \text{ if attribute k with specification l was absent in stimulus j} \end{cases}$$

The utility value of attribute k with specification l is then determined as the deviation of $\varphi_{k,\ l}$ from the average utility value over all stimuli (μ):

$$\beta_{k,l} = \varphi_{k,l} - \mu \tag{4}$$

$\beta_{k,l}$ = utility value of attribute k with specification l

As a last step, the relative importance, i.e., the weight, of an attribute k in the purchase decision is determined:

$$w_k = \frac{max_{l=1}^{L(k)}\left\{\beta_{k,l}\right\} - min_{l=1}^{L(k)}\left\{\beta_{k,l}\right\}}{\sum_{k=1}^{K}\left(max_{l=1}^{L(k)}\left\{\beta_{k,l}\right\} - min_{l=1}^{L(k)}\left\{\beta_{k,l}\right\}\right)} \tag{5}$$

w_k = weight of attribute k in the purchase decision

$L(k)$ = number of specifications for attribute k

K = number of attributes

It can be easily shown that sum of all weights w_k add up to one:

$$\sum_{k=1}^{K} w_k = 1 \tag{5.1}$$

The weight of an attribute k for the purchase decision is given in Eq. (5). For the following steps, the term $max_{l=1}^{L(k)}\left\{\beta_{k,l}\right\} - min_{l=1}^{L(k)}\left\{\beta_{k,l}\right\}$ is substituted by a_k because it is constant for a given k as it constitutes the span from the minimum to the maximum of the utility values ($\beta_{k,\ l}$) of attribute k. Therefore, we have:

$$w_k = \frac{a_k}{\sum_{k=1}^{K} a_k} \tag{5.2}$$

Adding up the weights of all attributes (w_k) leads to:

$$\sum_{k=1}^{K} w_k = \sum_{k=1}^{K} \frac{a_k}{\sum_{k=1}^{K} a_k} = \frac{\sum_{k=1}^{K} a_k}{\sum_{k=1}^{K} a_k} = 1 \tag{5.3}$$

3.2 Design of the case study

To be able to question several decision-makers about their implicit weighting of sustainability criteria, a limit conjoint analysis was set up that asked them to rank an exemplary vehicle component based on three attributes: cost (economic dimension), global warming potential (environmental dimension), and risk of social violation in the supply chain (social dimension). As the focus of this research laid on determining the general weighting of sustainability dimensions, i.e., inter-dimensional weighting, and not the weighting of individual indicators within a sustainability dimension, i.e., intra-dimensional weighting, the study was set up to contain one indicator that was seen a representative for each dimension by decision-makers. Cost, i.e., the cost to produce the respective component or procure it from a supplier, was chosen as a representative indicator for the economic dimension as it was deemed the most broadly used metric in regard to products and components. The global warming potential was chosen for the environmental dimension as this indicator was the most known and prominent metric for environmental life cycle impacts within the investigated company. Lastly, the risk of social violation in supply chains was chosen as the representative indicator for the social dimension as it did not focus on internal production processes but on the supply chain, which was known to decision-makers in connection with risk assessment of suppliers. “Social violations” in this analysis encompassed the decency of working conditions (e.g., no excessive working hours).

For every attribute, i.e., sustainability dimension, a low impact, medium impact, and high impact was defined. Table 1 shows the attributes and respective specifications. The concrete specifications were based on a gearbox which was chosen as a reference component. The impact of the gearbox was set as the medium impact in every sustainability dimension. According to Mettier et al. (2006; 2008), presenting relative changes to evaluate trade-offs is preferable as respondents cannot correctly interpret absolute impact values. Furthermore, pre-tests showed that respondents preferred to be presented with relative evaluation tasks. Another reason for the choice of a relative evaluation was the motivation to derive general implications of trade-off evaluations from decision-makers that could be transferred to other components. Therefore, the results from the study at hand can be used to derive implications for a company. These implications are presented in Section 4.2.

To derive the attribute specifications, the largest possible differences should be chosen that are still realistic (Green and Srinivasan 1978; Rao 2014). For the economic dimension, a change in price of up to 5% was found to be high but still realistic. Analogously, a change in GWP of up to 20% was chosen. The level of risks for social violations were semi-quantitative. Low risk indicated that the entire supply chain was known until the *n*-tier[1] supplier, i.e., the last supplier in the respective supply chain, no social violations were detected as of yet and due to the knowledge of the business practice of the respective suppliers also not to be expected. A medium level of risk indicated that the supply chain was not known until the *n*-tier supplier but risk assessment showed no risk of violation. A high risk denoted the possibility of unfair labor conditions in the supply chain of the component in question such as remuneration below the minimum wage of the respective industry sector or excessive working hours, i.e., weekly working hours exceeding 48 without compensation. The possibility of human rights violations, meaning e.g. possible child labor or forced labor in the supply chain, were not included in the analysis as they constituted a knockout (KO) criterion within the investigated automotive company, meaning that a possible supplier for that component with such a risk would immediately be rejected.

The possible combinations of the three selected attributes with three specifications each led to $3^3 = 27$ possible product alternatives, i.e., stimuli. As this study applied three properties with the same number of possible characteristics, the “Latin Square” was used to reduce the number of stimuli to a more manageable number that still represented the entire range of possible product alternatives (Hamlin 2005). This process led to a factorial design of nine stimuli. Figure 1 shows an exemplary stimulus card; the entire set of stimulus cards that were presented to the respondents can be found in the Electronic Supplementary Material (SM_1).

3.3 Study execution

Seventy decision-makers at an automotive company were asked to participate in the survey of which 54 agreed, which is a large enough sample to deduct hypotheses from (Orme 2010). The interview process with respondents was conducted as follows. The interview process was started off with an introduction about product sustainability and the motivation of

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[1] The term “*n*-tier” or “tier-*n*” is often used to express that a section in the supply chain beyond tier-1 (direct supplier) or tier-2 (sub-supplier) is concerned. The “*n*” symbolizes that it can be a position at variable depth in the supply chain, which also can differ in length—one supply chain might have five tiers while another might have eight. There is no consistent use of this term in the literature—it is used to denote the entire supply chain (Zimmer et al. 2017), parts of the supply chain after tier-1 or tier-2 (Wolf 2011), or to refer to the last supplier in a given supply chain (Wolf 2011; Kerkow et al. 2012). In this paper, “*n*-tier supplier” denotes the last supplier in a given supply chain.

Table 1 List of selected attributes and their specifications for the limit conjoint analysis

Attribute	Low	Medium	High
Cost	− 5%	Reference	+ 5%
Global warming potential	− 20%	Reference	+ 20%
Risk of social violation	Supply chain is known until *n*-tier; no violation to expect.	Reference: supply chain is not known until *n*-tier; still no violation to expect.	Supply chain is not known until *n*-tier; risk of unfair working conditions like low wage or excessive working time

the study. Then, the three sustainability dimensions and the respective indicators were explained while respondents were shown the nine stimulus cards. All respondents were shown the same set of stimulus cards (c.f. Electronic Supplementary Material SM_1). To put the relative changes into perspective, respondents were told the reference component (a gearbox) and the respective indicator values for the medium level of impact.

Respondents were then asked to envision that they could decide within their current function which of the presented alternatives would be used in a car and were asked to rank the stimulus cards in their order of preference. After completion of the ranking, the results, i.e., the respective weights for the sustainability dimensions, were presented to the respondent in order to review whether the results seemed plausible to the respondent. Afterwards, respondents were asked whether any attribute specification constituted a knockout (KO) criterion for them. Additionally, they were asked, whether sustainability was part of their expertise and daily business. It was specifically asked in that question if their efforts to consider sustainability included all three dimensions and whether they had a relation to the product. The interview process lasted between 20 and 30 min per respondent. Interviews were carried out with only one respondent at a time.

3.4 Evaluation and presentation of the results

In a first step, respondents who indicated that they viewed one or more attribute specifications as KO criteria were removed from the sample. The overall mean weight per sustainability dimension was derived by taking the arithmetic mean of all respondents. As an indicator of consistency of the sample, the mean squared error[2] (MSE) in relation to the overall mean was determined:

$$MSE=\frac{1}{n}\sum_{i=1}^{n}\left(\left(\overline{w_1}-w_{1,i}\right)^2+\left(\overline{w_2}-w_{2,i}\right)^2+\left(\overline{w_3}-w_{3,i}\right)^2\right) \qquad (6)$$

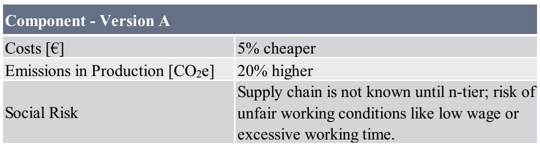
n = number of respondents$\overline{w_1}, w_{1,i}$ = overall arithmetic mean weight and weight of respondent i for the economic dimension, respectively$\overline{w_2}, w_{2,i}$ = overall arithmetic mean weight and weight of respondent i for the environmental dimension, respectively$\overline{w_3}, w_{3,i}$ = overall arithmetic mean weight and weight of respondent i for the social dimension, respectively.

The MSE was used as a relative measure, i.e., for determining the difference of the consistency within one cluster compared to the consistency within another cluster. Hence, a lower MSE of one cluster in comparison to the other indicated a higher consistency in the former cluster.

Depending on the organizational chart, respondents were chosen that represented the functions “Product Line,” “Product Management & Controlling,” “Production,” “Purchasing,” Research & Development,” “Sales,” and “Strategy & Communication.” Respondents of the “Product Line” cluster coordinated the inputs of the centers of competence for the development, purchasing, and production of a vehicle project. Due to data privacy concerns of the involved company, these seven functional clusters were anonymized and coded by number. In the following, “cluster” will always refer to the clustering by function. For each of these clusters, the mean weight per sustainability dimension was determined by taking the arithmetic mean of the respective group of respondents. As an indicator of consistency within each cluster, the mean squared error (MSE$_c$) of all respondents of the respective cluster in relation to the cluster mean was determined:

$$MSE_c=\frac{1}{n_c}\sum_{i=1}^{n_c}\left(\left(\hat{w}_{1,c}-w_{1,i,c}\right)^2+\left(\hat{w}_{1,c}-w_{2,i,c}\right)^2+\left(\hat{w}_{3,c}-w_{3,i,c}\right)^2\right) \qquad (7)$$

n_c = number of respondents in cluster c$\hat{w}_{1,c}, w_{1,i,c}$ = arithmetic mean weight of cluster c and weight of respondent i of cluster c for the economic dimension, respectively$\hat{w}_{2,c}, w_{2,i,c}$ = arithmetic mean weight of cluster c and weight of respondent i of cluster c for the environmental dimension,



Component - Version A	
Costs [€]	5% cheaper
Emissions in Production [CO2e]	20% higher
Social Risk	Supply chain is not known until n-tier; risk of unfair working conditions like low wage or excessive working time.

Fig. 1 Exemplary stimulus card

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[2] Even though the word “error” is contained in this metric, its application to measure consistency does not imply a judgment of any sort, but rather reflects the deviation from the average. It is not the intention to declare that respondents that deviate from the average are making an error.

respectively$\hat{w}_{3,c}, w_{3,i,c}$ = arithmetic mean weight of cluster c and weight of respondent i of cluster c for the social dimension, respectively

During the interview process, respondents were asked whether sustainability was part of their expertise and daily business. In the question, it was specified if their efforts to consider sustainability included all three dimensions and whether they had a relation to the product. If one of these conditions did not apply, respondents were classified as "Non-Sustainability" respondents. Thus, the sample could be divided into a "Sustainability" group, i.e., respondents who dealt with sustainability issues in their daily business, and a "Non-Sustainability" group, i.e., respondents who did not deal with sustainability issues in their daily business.

The last step of the analysis was to determine what effect outliers had on the results and, more specifically, what effect KO criteria had on the results as they are said to skew conjoint analysis outcomes. Therefore, the original sample ($n = 54$) was compared to differently adjusted versions of this sample. The first version ("KO") was purged from respondents who indicated that they viewed one or more attribute specifications as KO criterion (A). In a second version, the 20% of respondents with the highest mean squared error in regard to the overall arithmetic mean were declared as "top outliers" and removed (B). If a respondent accounted for more than 20% of the mean squared error regarding the individual respondent's distance to the respective cluster mean ("cluster outliers"), he or she was removed for a third version (C).

For every adjusted version of the sample, the overall mean weight per sustainability dimension was derived by taking the arithmetic mean of the remaining respondents. As an indicator of consistency of the entire sample, the MSE was calculated according to formula 6. The relative deviations of the adjusted samples to the original sample were determined as well as a mean deviation of weights of the sustainability dimensions. The mean deviation (Δ_k) for an adjusted sample was calculated as follows:

$$\Delta_k = \frac{\left|\Delta w_{1,k}\right| + \left|\Delta w_{2,k}\right| + \left|\Delta w_{3,k}\right|}{3} \tag{8}$$

$\Delta w_{1,k} = \frac{\overline{w}_{1,k}-\overline{w}_1}{\overline{w}_1}$ = relative difference of the mean weight for the economic dimension of the adjusted sample k to the original sample $\Delta w_{2,k} = \frac{\overline{w}_{2,k}-\overline{w}_2}{\overline{w}_2}$ = relative difference of the mean weight for the environmental dimension of the adjusted sample k to the original sample$\Delta w_{3,k} = \frac{\overline{w}_{3,k}-\overline{w}_3}{\overline{w}_3}$ = relative difference of the mean weight for the social dimension of the adjusted sample k to the original sample.

The absolute values of the relative differences were used in order to reflect their original order of magnitude in the resulting mean deviation.

For a joint presentation of the weights for the three dimensions, a triangle diagram was chosen (see Fig. 2). An introduction to triangle diagrams in relation to LCSA can be found in Hofstetter et al. (1999) or Finkbeiner et al. (2010). Every border of the triangle represents one sustainability dimension. In Fig. 2, the bottom left corner sees the economic dimension at 0%, the social dimension at 100%, and the environmental dimension at 0%. The further we move to the right on the bottom border, the higher the share of the economic dimension gets until it reaches the bottom right corner where the economic dimension accounts for 100%. To find the weight distribution of an individual point within the triangle, the respective guiding lines have to be followed to the respective border. The alignment of the axis labelling gives the indication which direction of the guiding lines pertains to which dimension. For example, the dark circle close to the bottom right corner in Fig. 2 represents a respondent who perceives the economic dimension as the most relevant one (weight of 71%), while assigning less weights for the social and environmental dimension (weight of 20 and 9%, respectively). The use of a triangle diagram is possible because the three weights always add up to 100%, as shown earlier (c.f. Eqs. (5.1)–(5.3)).

To be able to display the individual weights in a triangle diagram, they were translated into two-dimensional coordinates. The x and y coordinates for a respondent i were calculated as follows:

$$x_i = \frac{2\,w_{1,i} + w_{2,i}}{2} \tag{9}$$

$$y_i = \frac{\sqrt{3}\,w_{2,i}}{2} \tag{10}$$

The results of the case study and the respective triangle diagrams are presented in Section 4.1. To display the consistency of clusters, the MSE_c was plotted as a circle around the respective arithmetic cluster mean.

4 Results and discussion

4.1 Case study results

The individual weightings of the entire sample are shown in Fig. 2.

As can be seen from Fig. 2, there was no clear overall clustering visible. Even though a few respondents gave the exact same weight distribution (e.g., three respondents weighted the economic dimension at 21%, the environmental at 50% and the social at 29%—see where clusters 1, 2, and 4 coincide in Fig. 2), the individual weightings were clearly scattered. These results confirmed that deriving weights for product sustainability assessment is a challenging task (Finnveden 1997; Schmidt and Sullivan 2002; Finnveden et al. 2009; Cortés-Borda et al. 2013) as no clear picture

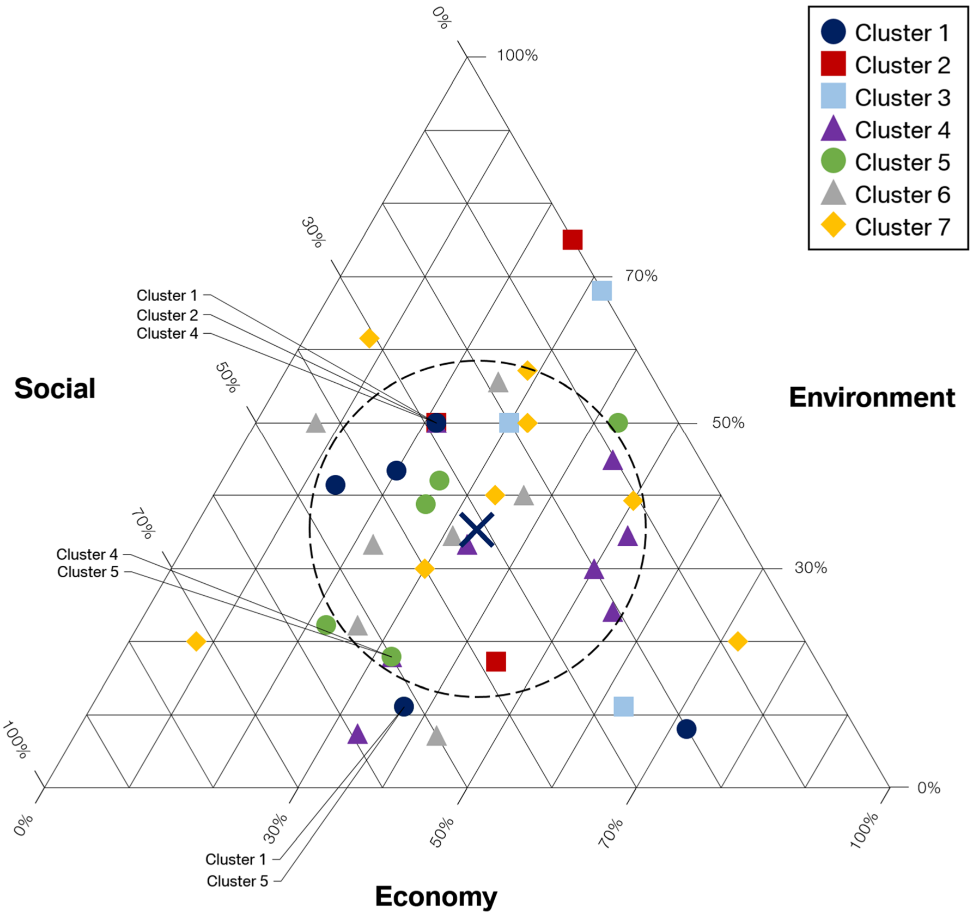


Fig. 2 Representation of all respondents ($n = 40$) and their individual weighting of sustainability dimensions. Additionally, the overall mean is given (**X**). The dotted line indicates the average distance of the individual respondent's weights to the overall mean (c.f. Section 3.4)

emerged even though decision-makers at the same company were asked.

The average weighted all three dimensions almost equally but placing more importance on the environmental dimension: the economic dimension was at 33.5%, the environmental at 35.2%, and the social at 31.2%. This still showed that all three dimensions were important for the overall sample when product decisions are considered. However, several respondents gave more extreme distributions of weight, e.g., two respondents weighted the social dimension at 0% (c.f. Fig. 2). These findings partly confirm the effect that Mettier et al. (2006; 2008) found, as conjoint analysis leads to more extreme values in elicited weights. However, while this held true on the individual level, the overall level arrived at a rather equal weighting.

Figure 3 shows the cluster means together with the overall mean. More detailed information on the cluster results are listed in Table 2. Exemplarily, cluster 4 is shown in more detail in Fig. 4.

As can be seen from Fig. 3 and Table 2, different function clusters had different weightings and different consistencies (c.f. MSE values). As the cluster with the highest MSE (cluster 7) was not the one with the least respondents and the one with the lowest MSE (cluster 6) was not the one with the most respondents, it appeared that consistency was not related to sample or cluster size. Overall, there was rather no consistent weighting within clusters (see Fig. 4 and Table 2). These two results may indicate that that weighting was more dependent on individual preference or assessment than related to the functional affiliation.

Cluster 3 constituted an exception to the other clusters as their focus was rather on process than product-related sustainability. That became apparent in their distance to the other cluster means (see Fig. 3). Cluster 2 placed a higher emphasis on the

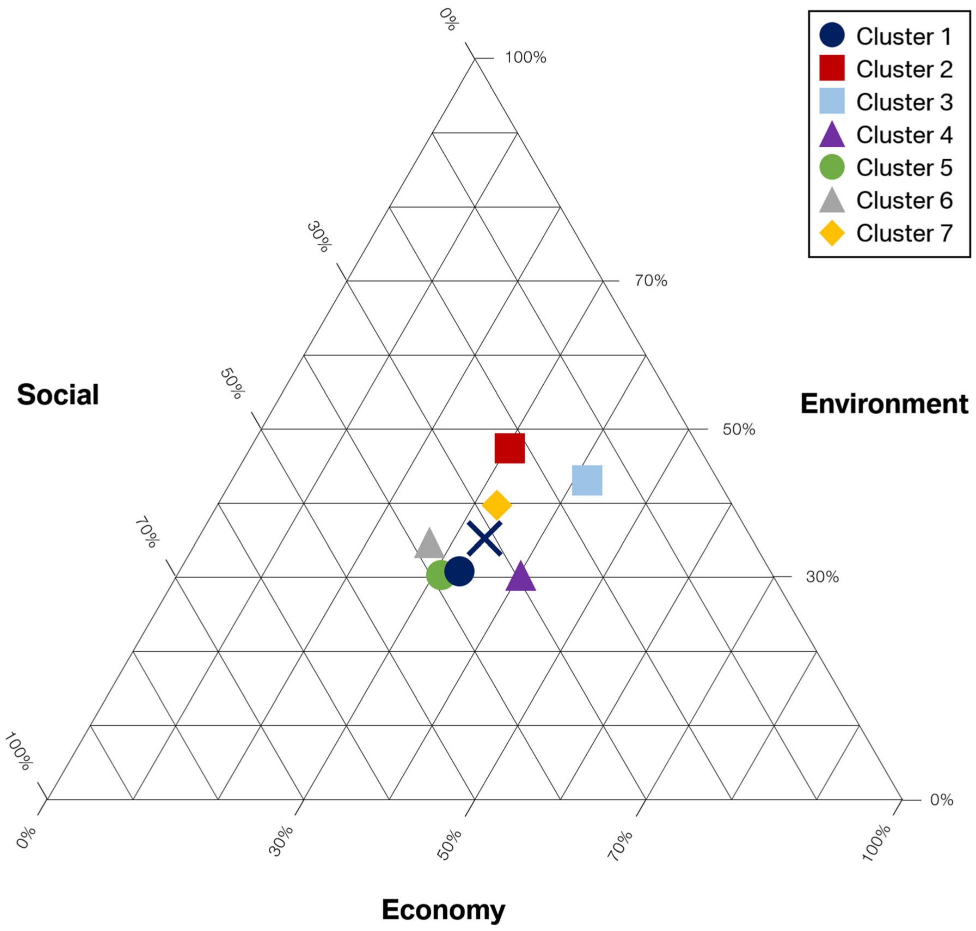


Fig. 3 Cluster means and overall mean (**X**)

environmental dimension than on the other clusters. This was related to a higher share of respondents working on sustainable products. The most consistent cluster was cluster 6, the "Sales" cluster, (MSE: 0.049) which weighed the social dimension stronger than the other sustainability dimensions. The low MSE implied that the respondents that were linked to the sales function had a rather consistent view on the importance of sustainability dimensions. Ideally, this view reflected the expectations of consumers or the market. The least consistency was observed in clusters 2 and 7. This might have to do with their broader array of responsibilities when compared to more "straightforward" functions like sales.

Table 2 Cluster results

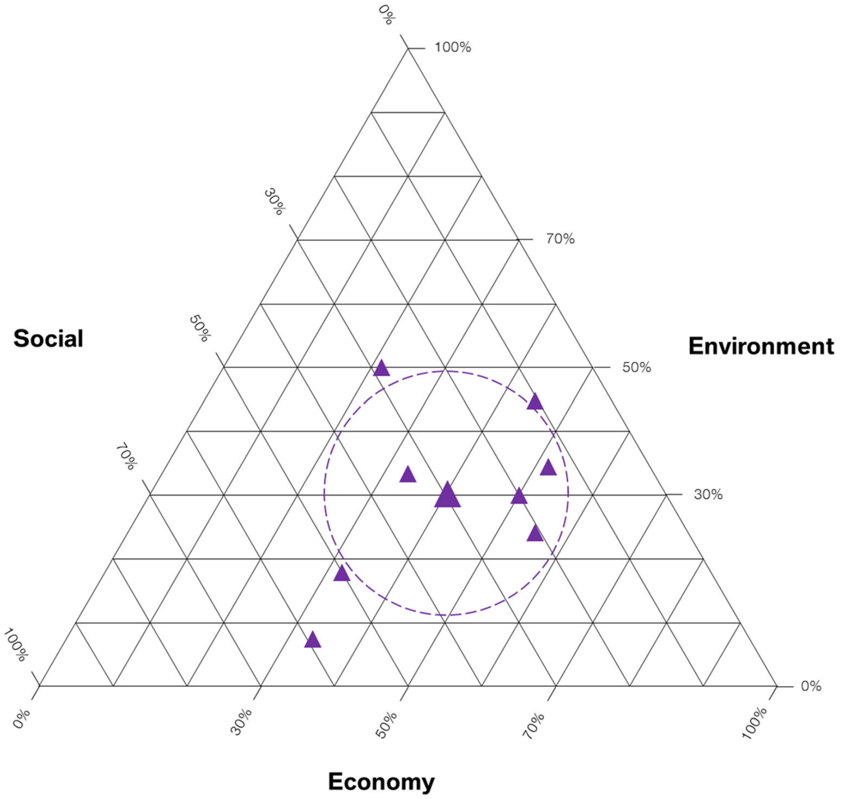
Cluster	Economy	Environment	Social	Respondents	MSE
1	32.9%	30.8%	36.4%	5	0.088
2	30.4%	47.4%	22.2%	3	0.093
3	41.7%	43.0%	15.3%	3	0.091
4	40.2%	30.3%	29.5%	8	0.055
5	31.0%	30.3%	38.7%	6	0.052
6	27.4%	34.7%	37.9%	7	0.049
7	32.7%	39.7%	27.5%	8	0.099
Total	33.5%	35.2%	31.2%	40	0.082

MSE mean squared error

Table 3 shows the results of the initial sample divided into sustainability and non-sustainability respondents.

Of the entire sample, 35% (14 of 40) were sustainability respondents. This share was over-representative as during the sampling process, it was ensured that enough sustainability respondents were included in order to be able to contrast a large enough group of sustainability respondents with a non-sustainability group. The sustainability respondents of the

Fig. 4 Cluster 4 showing a large spread in weights. The dotted line indicates the average distance of the individual respondent's weights to the cluster mean



entire sample weighted the environmental (37.6%) before the other two that were of equal importance. The non-sustainability respondents weighted the economic dimension (34.9%) before the environmental (34.0%) and social (31.1%) dimension. There were no sustainability respondents in clusters 3 and 6 of the evaluated sample. The strong emphasis of cluster 2 on the environmental dimension (Table 2) was carried by one sustainability respondent vs. two non-sustainability respondents. Especially clusters 1, 3, and 4 of non-sustainability respondents put strong weight on the economic dimension (> 40%). Unsurprisingly, it appeared that sustainability respondents paid more attention to the non-economical dimensions while non-sustainability respondents were primarily concerned with the economic dimension. However, non-sustainability respondents still regarded the environmental and social dimension with high importance.

Table 4 shows the overall mean of the original sample compared to the adjusted samples as described in Section 3.4.

Table 3 Weighting of the "sustainability" and "non-sustainability" groups by cluster of the entire sample

	Sustainability				Non-sustainability			
Cluster	Economy	Environment	Social	Respondents	Economy	Environment	Social	Respondents
1	17.6%	45.7%	36.7%	2	43.0%	20.8%	36.2%	3
2	25.0%	75.0%	0.0%	1	33.1%	33.6%	33.3%	2
3	–	–	–	0	41.7%	43.0%	15.3%	3
4	37.1%	35.7%	27.2%	4	43.4%	24.8%	31.8%	4
5	27.2%	20.0%	52.8%	2	32.9%	35.4%	31.7%	4
6	–	–	–	0	27.4%	34.7%	37.9%	7
7	34.1%	35.4%	30.4%	5	30.3%	46.9%	22.7%	3
Total	31.0%	37.6%	31.4%	14	34.9%	34.0%	31.1%	26

Table 4 Mean weights of the entire sample ("All") and the sample adjusted for KO criteria and outliers

Sample	Economy	Δ	Environment	Δ	Social	Δ	Respondents	Δ	Overall MSE	Δ	Mean deviation
All	37.0%	–	31.2%	–	31.8%	–	54	–	0.110	–	–
A (KO)	33.5%	−9%	35.2%	13%	31.2%	−2%	40	−26%	0.082	−26%	8%
B (top outliers)	34.9%	−6%	31.0%	0%	34.1%	7%	43	−20%	0.072	−35%	4%
C (cluster outliers)	32.5%	−12%	31.8%	2%	35.6%	12%	40	−26%	0.067	−39%	9%

Δ relative difference to "All"

Outliers did not appear to have had a big impact on the results of the initial sample as all three adjusted versions brought an average relative change in weights per sustainability dimension of < 10% (mean deviation in Table 4). The overall weighting shifted from economy to environment when removing KO (A). This was because several respondents declared that an increase in costs constituted a KO for them ($n = 6$). As expected, the adjusted sample showed a lower spread (MSE improved by up to 39%) and showed a more balanced distribution than the entire sample as the more extreme positions were removed. Interestingly, KO criteria did not seem to have as much an effect on results as outliers did. Version A showed an MSE that was higher than that of versions B and C even though these versions had the same or even higher number of respondents. It might be that respondents did report attributes as KO criteria when they saw them as highly important but did not really treat them as criteria to determine the rejection of a stimulus. In return, it might have occurred that respondents treated an attribute like a KO criterion but did not report it as such. Another possibility might be that subjective KO criteria do not skew conjoint analysis results as much as "regular" outliers. This would call into question the categorical removal of these respondents in Conjoint Analysis.

4.2 Implications for a company

The results of this research were derived from decision-makers by conducting a limit conjoint analysis based on a gearbox as a reference component. Therefore, the derived weights hold true for that component. However, as the stimulus cards were phrased to evaluate relative trade-offs, the argument of this paper is that the results simultaneously can give an indication on the respondents' decision for other components as well. The budget for the development and production of vehicle components is allocated on a relative basis, meaning that expensive components get a higher budget than cheap ones. An absolute increase of one Euro production cost is not as critical for an expensive component (like a gearbox, which ranges above one hundred Euros) as it would be for a cheap one (like a screw, which ranges below one Euro). While an increase of one Euro would still be accepted for the expensive component, the same cost increase for the cheap component would definitely be rejected. The evaluation of absolute changes in cost can therefore not be transferred from one component to another. The relative evaluation, however, can be transferred to other components within a car as it reflects the general evaluation of trade-offs that also remain valid when the absolute basis, which for a car usually does not span more than three orders of magnitude for the evaluated indicators, for evaluation changes. Therefore, it was deemed plausible to look into what implications for companies could be derived if these weights are generalized to an extent and seen as valid for other components.

It has been shown that, overall, decision-makers at an automotive company considered all three sustainability dimensions almost equally. The derived set of weights could be used to aggregate LCSA results based on the presented indicators to support decision-making based on the now quantified decision-makers' preference that already exists. The question of interest for a sustainability expert at an industry company looking to implement decision support based on LCSA results would be regarding the concrete suggestions derived from this research. To determine whether the derived distribution of weights between sustainability dimensions would be company-optimal, the expectations of consumers are of interest. For this research, the weighting of the "Sales" cluster (cluster 6) was taken as proxy for consumer expectation, which means that the ideal weight distribution would give 37.9% to the social, 34.7% to the environmental and 27.4% to the economic dimension (see Table 2). Therefore, the observed overall weighting of sustainability dimensions at the investigated company came fairly close to that optimal weighting in the environmental dimension but had a tilt towards the economic dimension. However, even though the overall weighting met consumer expectations, individual weightings were far off in some cases.

Practitioners might now see two possible weighting sets to implement in an LCSA framework based on the presented results:

1. Adopt the overall mean weighting of the evaluated sample
2. Adopt the weighting of cluster 6 (corresponding to the consumer expectation)

This research would suggest following the first path as that weighting set reflected the combined decision-making process within the respective company and was based on the entire sample. Apart from the higher sample size, the fact that it was a quantification of the already existing decision process within the company is expected to increase its acceptance for adoption within the concerned company. Obviously, this logic is only valid for the evaluated company. Practitioners seeking to find the optimal weighting set for putting LCSA into practice at their company would have to investigate the overall weighting of decision-makers at the respective company first.

Additionally, this research showed that the following points influence the weight distribution in the decision-making process at a company:

- When multiple decision-makers are included in the decision process, the overall weighting came close to an equal weighting of all three dimensions, neutralizing extreme individual weightings.
- Function seemed to have less of an impact on weighting as individual preference weighting according to this research.
- Sustainability decision-makers put more weight on the environmental and social dimension as opposed to the non-sustainability decision-makers.

As individual preference or assessment seemed to be more dominant than function in the weighting of sustainability dimensions, the driving forces of individual assessment would be of interest to gain a better understanding of how to ensure balanced weighting in the decision process.

4.3 Limitations and future research needs

The main limitations of this research laid in its sample size, the focus on one reference component, the geographical scope and the evaluation of just the three overarching sustainability dimensions. Future research could look at expanding the sample size within a company or across companies and to carry out studies looking at other components or products. In addition, the weight distributions in other countries than Germany would be of interest. To further detail the weighting in LCSA-based decision-making, the weighting of indicators within one sustainability dimension should be looked at, for example, it would be of interest how to weight GWP against land use or resource use. The implications for practitioners were derived based on the assumptions that the analysis results could be generalized and that the “Sales” cluster of the investigated sample could be taken as proxy for the consumer’s expectation. Future research could therefore test the weighting for different components to confirm whether results can be generalist. For consumer expectation, a similar study questioning consumers directly could be carried out to gather their weight distribution. Furthermore, the drivers that influence personal weighting of sustainability dimensions would be a promising field of research. In relation to that question, it would be interesting to see whether function really has no significant influence on the weighting of sustainability dimensions as implied in this research.

The limit conjoint analysis also comes with a limitation in itself. As the evaluation of the ranking sees distances between ranks as being equal, there is no way of capturing, whether respondents see large or small differences between certain stimuli. Furthermore, as it had been shown that KO criteria did not have as strongly an effect on overall results as outliers did, the additional consideration of responses of “KO”-respondents might yield additional insights.

5 Conclusions

The goal of the paper was to determine weights for sustainability assessment of products at an automotive company to support decision-making in an operationalized LCSA framework by offering a way to solve a multi-criteria decision issue. The applied method to derive weights was limit conjoint analysis that asked the (implicit) weights of 54 managers at an automotive company via ranking of nine alternatives of a reference component that differed in their performance in regard to the three sustainability dimensions. The results of the sample without respondents that saw KO criteria in any of the stimuli ($n = 40$) showed a large spread in weighting without clear clustering. This confirmed that weighting in product sustainability assessment is a challenge as no clear picture emerged even when decision-makers at the same company were asked. On average, all sustainability dimensions were considered almost equally with the environmental dimension being slightly more important. Singular respondents applied more extreme distributions of weight. Nevertheless, results were not significantly impacted by extreme individual weighing as adjusting for KO criteria and outliers changed weights on average by less than 10%. Function clusters showed low consistency in themselves hinting that weighting was more of a personal than a functional issue. Sustainability respondents (35% of the sample) weighted the social before the environmental and economic dimensions while non-sustainability respondents put the economic dimension before the other two. Interestingly, KO criteria did not influence results as much as outliers did.

To derive practical implications for introducing weights for sustainability dimensions at a company, it was assumed that the results of the analysis could be generalized and that the ideal weighting set would correspond to the customers’ expectations. When company-optimal weighting was guided by consumer expectation (the weighting of the “Sales” cluster was taken as proxy for consumer expectation), the social dimension should be considered before the environmental and the economic dimension. From the results, two possible implications for the adoption of decision-making support at a company based on LCSA could be derived: either the adoption of the overall mean weighting of the investigated sample could be adopted or the weighting set of the "Sales" cluster could be adopted to consider the consumer's expectation. The recommendation would be to follow the adoption of the first route as the outcome of the overall sample represented a quantification of the already existing decision process. Thus, the acceptance of the first route is expected to be higher.

It could be shown that Conjoint Analysis enabled decision-making within LCSA by introducing weights to solve an MCDA issue. Furthermore, implications for practitioners could be derived to ensure company-optimal decision-making related to product sustainability. Future research should look at expanding the sample size, evaluate different reference components or products and widen the geographical scope. Furthermore, investigating the weighting of indicators within sustainability dimensions and the drivers that influence personal preferences and decision-making in regard to weighting sustainability dimensions could be of interest for future research.

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References

Adepoju JA, Ipinyomi RA (2016) Construction of asymmetric fractional factorial designs. Int J Eng Appl Sci 3:88–91

Alfares HK, Duffuaa SO (2008) Assigning cardinal weights in multi-criteria decision making based on ordinal ranking. J Multi-Criteria Decis Anal 15:125–133

Alriksson S, Öberg T (2008) Conjoint analysis for environmental evaluation. Environ Sci Pollut Res 15:244–257

Backhaus K, Erichson B, Plinke W, Weiber R (2011a) Multivariate Analysemethoden. Eine anwendungsorientierte Einführung, 13th edn. Springer-Verlag, Berlin

Backhaus K, Erichson B, Weiber R (2011b) Fortgeschrittene Multivariate Analysemethoden. Eine anwendungsorientierte Einführung., 1st edn. Springer-Verlag, Berlin

Baier D, Brusch M (eds) (2009) Conjointanalyse. Springer Berlin Heidelberg, Berlin

Bond A, Morrison-Saunders A, Pope J (2012) Sustainability assessment: the state of the art. Impact Assess Proj Apprais 30:53–62

Cattin P, Wittink DR (1982) Commercial use of conjoint analysis: a survey. J Mark 46:44–53

Choo EU, Schoner B, Wedley WC (1999) Interpretation of criteria weights in multicriteria decision making. Comput Ind Eng 37:527–541

Cortés-Borda D, Guillén-Gosálbez G, Esteller LJ (2013) On the use of weighting in LCA: translating decision makers' preferences into weights via linear programming. Int J Life Cycle Assess 18:948–957

Diakoulaki D, Mavrotas G, Papayannakis L (1995) Determining objective weights in multiple criteria problems: the critic method. Comput Oper Res 22:763–770

Figueira J, Roy B (2002) Determining the weights of criteria in the ELECTRE type methods with a revised Simos' procedure. Eur J Oper Res 139:317–326

Finkbeiner M, Reimann K, Ackermann R (2008) Life cycle sustainability assessment (LCSA) for products and processes. In: SETAC Europe 18th annual meeting, 25–29 May. Warsaw, Poland

Finkbeiner M, Schau EM, Lehmann A, Traverso M (2010) Towards life cycle sustainability assessment. Sustainability 2:3309–3322

Finnveden G (1997) Valuation methods within LCA—where are the values? Int J Life Cycle Assess 2:163–169

Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S, Koehler A, Pennington D, Suh S (2009) Recent developments in life cycle assessment. J Environ Manag 91:1–21

Green PE, Srinivasan V (1978) Conjoint analysis in consumer research: issues and outlook. J Consum Res 5:103

Hamlin RP (2005) The rise and fall of the Latin Square in marketing: a cautionary tale. Eur J Mark 39:328–350

Heijungs R, Huppes G, Guinée JB (2010) Life cycle assessment and sustainability analysis of products, materials and technologies. Toward a scientific framework for sustainability life cycle analysis. Polym Degrad Stab 95:422–428

Hofstetter P, Braunschweig A, Mettier T, Müller-Wenk R, Tietje O (1999) The mixing triangle: correlation and graphical decision support for LCA-based comparisons. J Ind Ecol 3:97–115

Itsubo N, Inaba A (2003) A new LCIA method: LIME has been completed. Int J Life Cycle Assess 8:305

Itsubo N, Sakagami M, Washida T, Kokubu K, Inaba A (2004) Weighting across safeguard subjects for LCIA through the application of conjoint analysis. Int J Life Cycle Assess 9:196–205

Itsubo N, Sakagami M, Kuriyama K, Inaba A (2012) Statistical analysis for the development of national average weighting factors—visualization of the variability between each individual's environmental thoughts. Int J Life Cycle Assess 17:488–498

Johnson R, Orme B (1996) How many questions should you ask in choice-based conjoint studies? Sawtooth Softw Research P:23

JRC (2012) Towards a life-cycle based European sustainability footprint framework. Publications Office of the European Union, Luxembourg

Kerkow U, Martens J, Müller A (2012) Vom Erz zum Auto - Abbaubedingungen und Lieferketten im Rohstoffsektor und die Verantwortung der deutschen Automobilindustrie

Klöpffer W (2003) Life-cycle based methods for sustainable product development. Int J Life Cycle Assess 8:157–159

Klöpffer W (2008) Life cycle sustainability assessment of products. Int J Life Cycle Assess 13:89–95

Klöpffer W, Grahl B (2014) From LCA to sustainability assessment. In: Life Cycle Assessment (LCA). Wiley-VCH Verlag GmbH & Co. KGaA, Weinheim, pp 357–374

Kohne F, Totz C, Wehmeyer K (2005) Consumer preferences for location-based service attributes: a conjoint analysis. Int J Manag Decis Mak 6:16

Mettier TM, Hofstetter P (2004) Survey insights into weighting environmental damages: influence of context and group. J Ind Ecol 8:189–209

Mettier T, Scholz RW (2008) Measuring preferences on environmental damages in LCIA. Part 2: choice and allocation questions in panel methods. Int J Life Cycle Assess 13:468–476

Mettier T, Scholz R, Tietje O (2006) Measuring preferences on environmental damages in LCIA. Part 1: cognitive limits in panel surveys (9 pp). Int J Life Cycle Assess 11:394–402

Orme B (2010) Sample size issues for conjoint analysis. In: Getting started with conjoint analysis: strategies for product design and pricing research, second Edi. Research Publishers LLC, Madison, pp 57–66

Ostermeyer Y, Wallbaum H, Reuter F (2013) Multidimensional Pareto optimization as an approach for site-specific building refurbishment solutions applicable for life cycle sustainability assessment. Int J Life Cycle Assess 18:1762–1779

Rao VR (2014) Theory and design of conjoint studies (ratings based methods). In: Applied conjoint analysis. Springer Berlin Heidelberg, Berlin, Heidelberg, pp 37–78

Schmidt W-P, Sullivan J (2002) Weighting in life cycle assessments in a global context. Int J Life Cycle Assess 7:5–10

Scholl A, Manthey L, Helm R, Steiner M (2005) Solving multiattribute design problems with analytic hierarchy process and conjoint analysis: an empirical comparison. Eur J Oper Res 164:760–777

Sichtmann C, Stingel S (2007) Limit conjoint analysis and Vickrey auction as methods to elicit consumers' willingness-to-pay. Eur J Mark 41:1359–1374

Singh RK, Murty HR, Gupta SK, Dikshit AK (2009) An overview of sustainability assessment methodologies. Ecol Indic 15:281–299

Skiera B, Gensler S (2002) Berechnung von Nutzenfunktionen und Marktsimulationen mit Hilfe der Conjoint-Analyse, Teil I. Wirtschaftswissenschaftliches Stud 31(4):200–206

Tarne P, Traverso M, Finkbeiner M (2017) Review of life cycle sustainability assessment and potential for its adoption at an automotive company. Sustainability 9:670

Traverso M, Finkbeiner M, Jørgensen A, Schneider L (2012) Life cycle sustainability dashboard. J Ind Ecol 16:680–688

Traverso M, Tarne P, Wagner V (2015) Towards a comprehensive approach for the sustainability assessment of a product: product social impact assessment. In: Pfeffer P (ed) 6th International Munich Chassis Symposium 2015. Springer-Verlag, pp 161–174

UNEP/SETAC Life Cycle Initiative (2011) Towards a life cycle sustainability assessment - making informed choices on products. UNEP/SETAC Life Cycle Initiative, France

Voeth M (2000) Nutzenmessung in der Kaufverhaltensforschung. Deutscher Universitätsverlag, Wiesbaden

Voeth M, Hahn C (1998) Limit conjoint-analyse. Mark Zeitschrift für Forsch und Prax 20:119–132

Wittink DR, Cattin P (1989) Commercial use of conjoint analysis: an update. J Mark 53:91–96

Wolf J (2011) Sustainable supply chain management integration: a qualitative analysis of the German manufacturing industry. J Bus Ethics 102:221–235

Yang G, Yang J-B, Xu D-L, Khoveyni M (2017) A three-stage hybrid approach for weight assignment in MADM. Omega 71:93–105

Zimmer K, Fröhling M, Breun P, Schultmann F (2017) Assessing social risks of global supply chains: a quantitative analytical approach and its application to supplier selection in the German automotive industry. J Clean Prod 149:96–109

2.3 Mechanism to Implement LCSA Results in Decision Process

In the previous sections, the assessment of the S-LCA dimension within LCSA and the combination of LCA, LCC and S-LCA results to an overall LCSA criticality score have been addressed. This section deals with the third research objective (c.f. section 1.5) and presents the results of a publication by Tarne et al. (2018) on monetizing sustainability impacts. The study investigated how decisions for more sustainable product alternatives could be supported within a company by introducing a Product Sustainability Budget (PSB) that would cover potential increases in cost of the more sustainable product. The study investigated the preferences of 250 potential premium car buyers represented by a sustainability interior package. Choice-Based Conjoint Analysis (CBCA) was applied to determine the utility value of this sustainability package in relation to other vehicle properties such as cost, fuel consumption, drive train and engine power. To obtain the willingness to pay of customers for the sustainability package, an advanced Van-Westendorp Analysis was combined with the CBCA. Results showed that 19% of the investigated sample would prefer the sustainability interior package which could be sold.

The paper addressed the research objective 3 (c.f. section 1.5) in the following way:

- **Development of a method of monetizing product sustainability impacts** It could be shown that by applying a monetization approach to measure the willingness to pay (WTP) of consumers for directly perceivable product features, a business case for a sustainability interior package for a car can be made. The study combined CBCA to identify a target group within a sample for the German premium car segment that would prefer the sustainability package to other interior packages and consider it in a purchase decision.
- **Devise a mechanism to integrate findings in LCSA framework at an automotive company** The proposed PSB would divert part of this profit towards measures for improving product sustainability. The PSB would function as a solution space, i.e. a range in which costs could rise without having an influence on the economic dimension of the LCSA impacts. The creation of this solution space enabled finding the most effective set of measures for improving overall product sustainability,

The publication by Tarne et al. (2018) is presented on pages 86–105.

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In this section it was shown how the investigation of the customer added value of product sustainability features could be used to create a mechanism — the PSB — that enables the identification of the most effective set of measures for improving overall product sustainability despite a potential increase in costs. How the PSB is used to implement the LCSA results in the product-related decision process is shown in the next section as well as how the combination of the individual findings of sections 2.1–2.3 contribute to achieving the overarching goal of this thesis.

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LIFE CYCLE SUSTAINABILITY ASSESSMENT

Introducing a product sustainability budget at an automotive company—one option to increase the use of LCSA results in decision-making processes

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Abstract

Purpose Product sustainability assessment should evaluate the impacts on all three dimensions of sustainability (environment, economy, and society). Life cycle sustainability assessment (LCSA) is a framework that extends life cycle-based product assessment to all three dimensions. Evaluation of trade-off situations poses a challenge within LCSA in a business context, especially if improvement measures for product sustainability lead to higher costs. This paper introduces the concept of the Product Sustainability Budget (PSB) to enable a decision for improvement measures despite of rising costs. It demonstrates a way to create such a PSB and how to combine it with an operationalized LCSA framework at an automotive company.

Methods A survey was carried out asking 250 potential customers of the premium car segment in Germany via Choice-Based-Conjoint-Analysis (CBCA) about their preference of a sustainability interior package in a car. The sustainability package was one of the three specifications of a potential car interior (standard, luxury, sustainability) and was asked along four other attributes (price, drive train, engine power, and consumption). The survey was expanded by an Advanced-Van-Westendorp analysis to ask respondents about their willingness-to-pay (WTP) for such a package. The major findings of the study (take rate and price for the sustainability interior package) were then implemented in a business case logic from which the PSB was created.

Results and discussion Nineteen percent of the entire sample would prefer the sustainability interior package to the other packages (=potential take rate) while the rest (81%) favored the luxury package. The package should be sold to this (potential) target group at price premium of 1.3–1.7% for a middle class limousine (or 0.4–1.1% when corrected for overstated WTP). It could be shown in a theoretical business case logic for such a sustainability package that the profit could be converted to form the PSB, which could compensate an increase in costs caused by a measure to improve product sustainability. The PSB opened up a solution space to identify the ideal set out of several possible improvement measures.

Conclusions The introduction of an LCSA evaluation scheme on component level in combination with the proposed Product Sustainability Budget could enable substantial product sustainability improvement even when costs increase. The combination of an implicit CBCA and an explicit WTP study delivered a sound basis for creating this Product Sustainability Budget. The proposed concept should be applied in a business context to test its viability and additionally investigate customers' WTP for improved social impacts.

Keywords Advanced-Van-Westendorp analysis · CBCA · Choice-based conjoint analysis · Life cycle sustainability assessment · Monetary valuation · Monetization · Willingness-to-pay · WTP

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1 Introduction and goals

Life cycle sustainability assessment (LCSA) assesses the impacts of a product over its entire life cycle on all three dimensions of sustainability (Klöpffer 2003; Finkbeiner et al. 2008; Klöpffer 2008; Finkbeiner et al. 2010). The operationalization of this framework should enable decision makers to improve product sustainability impacts in a business context. There are different approaches how to evaluate impacts on the three sustainability dimensions (economy, environment, and society) in order to achieve sustainability. They can be grouped into promoting "strong" or "weak" sustainability (JRC 2012). Within the approaches applying the concept of weak sustainability, improvements of impacts on one dimension can compensate for deterioration of impacts on other dimensions (Singh et al. 2009). The approaches that apply the concept of strong sustainability adhere to a hierarchical treatment of the dimensions allowing for no compensation. Within strong sustainability, the environmental dimension constitutes the highest instance as all social and economic interactions take place within the boundaries of our planet. The social dimension then constitutes the limits for economic activities (Singh et al. 2009; JRC 2012). In this way, the concept of strong sustainability respects humanity's "limits to growth" (Meadows et al. 1972; Meadows et al. 2007a). However, the discussion around sustainable development in the business world, most notably the World Business Council For Sustainable Development (WBCSD), is more geared towards the weak interpretation of sustainability (Davies 2013).

When presented with LCSA results, decision makers have to make decisions based on multiple criteria when evaluating measures or product concepts that differently affect the three dimensions of sustainability. This poses a challenge to finding an optimal solution. On the one hand, it is not clear how to deal with trade-offs between sustainability dimensions making it harder to come to a clear decision. On the other hand, when impacts on the economic dimension are concerned (e.g., higher costs due to a switch to a more sustainable production technique), a product alternative with higher costs might not even be considered because the economic performance of products is usually pivotal at companies that are themselves evaluated based on their economic performance. The latter challenge could be addressed by applying monetary valuation (Tarne et al. 2017). The method of monetary valuation tries to translate the non-monetary impacts on the environmental and social dimension into monetary terms. This would bring impacts on all sustainability dimensions to the same level and make them easily comparable, eliminating a dominant influence of the economic dimension and making trade-offs easier to interpret.

To address the challenge of enabling decisions despite increasing costs combined with the operationalization of LCSA, the authors propose the introduction of the Product Sustainability Budget (PSB). This budget is a new concept created and introduced in this paper to translate the customer value of product sustainability features to create a solution space to find the ideal solution within a business context for the improvement of product sustainability based on LCSA evaluation. The PSB functions as complement to the LCSA framework. It helps to support the decision-making process on the improvement actions to be undertaken as indicated by LCSA, under the conditions of costs constraints. The objective of the PSB is to enhance the operability of LCSA, i.e., increase the usability of its results, rather than finding a differentiated way of monetizing the impacts on all three sustainability dimensions.

The goal of this paper is to develop and demonstrate a way to create this Product Sustainability Budget. The budget should be able to be used in combination with an LCSA assessment approach to find the optimal product concept or improvement solution for the product's sustainability performance. It should also enable the decision for product concepts or improvement measures that are an overall improvement of sustainability performance according to LCSA but would contribute to the increase of production cost.

To achieve this goal, the existing approaches to monetary valuation were looked at (Sect. 2.1) and the most relevant for the operationalization of LCSA within a business context was chosen (Sect. 2.2). Then, a specific method of monetary valuation within the chosen approach was selected (Sect. 2.3). A way to put LCSA into practice at an automotive company was introduced in Sect. 2.4. The study design and setup to determine the monetary value of sustainability impacts are shown in Sect. 3.1. The link between the study results and the LCSA framework by the creation of the Product Sustainability Budget is presented in Sect. 3.2.

The contribution of this paper is twofold. The first contribution is a concrete study regarding the evaluation of willingness-to-pay (WTP) for sustainability in the interior of a car. The second contribution is of conceptual nature, showing how these concrete results could be used to create the Product Sustainability Budget, which enables decision support based on LCSA results when evaluated improvement measures or product alternatives lead to higher costs.

2 Background

In this section, a short overview of monetary valuation approaches is given (Sect. 2.1). The successive selection of an approach for monetary valuation within this research is presented afterwards. It started with the decision for one approach out of the existing monetary valuation approaches (Sect. 2.2). The decision for a specific monetary valuation method within the chosen approach is presented in Sect. 2.3. The link of the monetary valuation with the LCSA framework aiming at facilitating LCSA-based decision making at an automotive company is introduced in Sect. 2.4.

2.1 Approaches to monetary valuation

Approaches to monetary valuation focus either on impacts on society or a company. Several methods or combinations of methods are applied within these approaches to determine what monetary values certain environmental impacts have. It is worth noting that different approaches often resort to the same methods, e.g., the Natural Capital Protocol as well as the ExternE approach used the WTP method to determine monetary evaluation.

The majority of approaches to determine impacts on society deals with the evaluation of costs to society due to environmental impacts. In general, the quantification of externalities, i.e., costs not represented in the market, falls into this category. A prominent project that attempted the internalization of externalities was the ExternE project (Markandya 2012) which developed an approach to determine externalities from electricity generation. They proposed an impact pathway approach that modeled the generation of emissions, their dispersion, the resulting impact on humans, and finally the monetary valuation of those impacts. The monetary valuation was done by determining the willingness-to-pay (WTP) to avoid the resulting impacts (European Commission 2005). Another approach to the monetary valuation of environmental impacts that got attention was PUMA's Environmental Profit and Loss (EP&L) project (PUMA 2011). The company set out to determine the positive and negative externalities that their production activities and supply chains generated. The "Sustainable Value" approach (Figge and Hahn 2004; Figge and Hahn 2005; Hahn et al. 2013) is also often found in relation to monetary valuation even though it would more correctly be classified as an "efficiency" approach. The approach proposes to evaluate companies based on key performance indicators pertaining to the three sustainability dimensions (e.g., profit, CO_2 emissions, number of employees) and build an industry average based on these evaluations. The sustainability assessment model (SAM) (Baxter et al. 2003; Bebbington et al. 2007; Frame and Cavanagh 2009) is sometimes also related to monetary valuation even though it "retains a qualified commitment to monetization" (Bebbington et al. 2007).

The Natural Capital Protocol (NCP) by the Natural Capital Coalition gives extensive guidance to businesses as to how to meaningfully include valuation of environmental impacts or ecosystem services into their business accounting (Natural Capital Coalition 2016). Another approach was taken by Biesalski and Co. (2014) who determined the extent to which the sustainability image contributes to the entire brand value. In that way, companies can gauge the (indirect) monetary effect that negative or positive environmental performance could have on the company.

2.2 Choice and focus of monetary valuation approach

This research intends to identify an approach relevant for the operationalization of LCSA at a company. Therefore, the approaches focusing on society were not considered. From the approaches relating to a company's viewpoint, focusing on the benefits of improved product sustainability to differentiate the product from competitors, as proposed by the NCP, was chosen. This approach was deemed to be more promising for the promotion of the benefits of product sustainability improvements within a company.

2.2.1 Effects of monetary valuation of sustainability dimensions

Monetary valuation of sustainability impacts transforms qualitative information into monetary (quantitative) units. Monetary valuation of social impacts therefore enables the compensation of drastic individual impacts, for instance inhuman labor conditions like enslavement and death of workers, as they are assigned a cost. That means that they could be compensated by increased revenues or improved environmental performance. In some cases, this could lead to decisions for very profitable but inhumane product alternatives. Thus, monetary valuation of social impacts might support the violation of human rights and is therefore not applied in this research.

When attempting the monetary valuation of environmental impacts, a few points need to be considered. Currently, monetary valuations of environmental life cycle impacts are mainly carried out in relation to the societal costs (c.f. ExternE, EP&L etc.) that are not yet directly relevant for companies unless regulators mandate the internalization of these costs (Stanton 2012; Natural Capital Coalition 2016). To express the company relevance of product sustainability performance, the route of determining the benefit for the company by finding out the added value for the customer was chosen for this research. That means that the Product Sustainability Budget in this research does not perform a monetary valuation of the impact category results of an LCSA and its determined impacts on all three sustainability dimensions directly. Instead, the customer value of sustainability features in a product is determined and later linked to the LCSA framework. That way, an amount to which additional costs do not affect the economic dimension within the operationalized LCSA framework (c.f. Sect. 2.4) can be set.

2.2.2 Customer relevant aspects

Groening et al. (2015) found that customers are mainly interested in the emissions pertaining to the use of a vehicle as they fall in their realm of responsibility. Emissions related to the product's manufacture are the least important to them (Groening et al. 2015). However, as the CO_2 emissions of the use phase are already regulated by the European Commission for the European market (European Commission 2007), this research laid its focus on the customer relevant aspects of the product manufacture. Customers are only interested in improved sustainability impacts when they are coupled with increased performance or increased customer satisfaction (Ottman et al. 2006; Biswas 2016). Thus, this research focused on determining the customer value of sustainable interior components rather than the monetary value of the impacts of the vehicle's production or its components.

2.3 Specific monetary valuation method

When it comes to determining the added value of product features for customers, WTP is considered to be “a monolith” (Lankoski 2010). This section gives an overview of WTP studies related to product sustainability impacts (Sect. 2.3.1) and lays out the selection of a specific WTP method (Sect. 2.3.2). Subsequently, the background for the chosen methods, Choice-Based-Conjoint analysis (Sect. 2.3.3) and Advanced-Van-Westendorp analysis (Sect. 2.3.4) are presented.

2.3.1 Previous studies

There are many studies on the consumer added value or their willingness-to-pay a price premium for product sustainability or its influence on the purchase decision. Depending on the study focus and product group, the results vary but generally indicate that product sustainability constitutes added value for consumers that can be capitalized.

Meta-studies A meta-study by Tully and Winer (2013) of 83 WTP studies found that 60% of respondents were willing to pay an average price premium of ca. 17% for socially responsible products. The premium was lower for durable than non-durable goods (Tully and Winer 2013). Cai and Aguilar (2013) found a wide range of WTP in their meta-study of 19 studies on environmentally certified wood products. According to their analysis, customers were willing to pay between 1% and ca. 40% more for certified products with low-end and frequently used wood products being more likely to yield premiums (Cai and Aguilar 2013). The meta-analysis of 18 studies on the WTP for green electricity by Sundt and Rehdanz (2015) showed that information on the effects of green electricity increased the WTP of consumers.

Various products A study by the European Commission of nearly 27,000 respondents found that 75% were willing to buy environmentally friendly products even if they were more expensive than conventional ones (European Commission 2008). In 2013, they further determined that 77% of ca. 25,000 respondents were willing to pay up to 5% more for not further specified sustainable products (European Commission 2013). Further studies found various WTP for different products, e.g., 12% price premium for pet food by German consumers (Völker and Tachkov 2013), ca. 16% price premium for organic shirts by half of the interviewed US consumers (Ha-Brookshire and Norum 2011) or 2–16% price premium for environmentally labeled furniture in England and Sweden (Veisten 2007). However, in the luxury goods sector, environmental concerns are not always desired by customers: Achabou and Dekhili (2013) found that French consumers did not desire recycled contents in luxury clothing.

WTP for automobiles Consumers rate financial factors before environmental ones (Krupa et al. 2014). Even with expected fuel cost savings when using a hybrid vehicle, consumers are only willing to pay a modest premium which often lies under the initial premium that car manufacturers demand for hybrid vehicles (Krupa et al. 2014; Liu 2014). For German consumers, the focus still lies on the use phase and their WTP for a decrease of CO_2 emissions is positive but dependent on their sociographic affiliation (Achtnicht 2012). Hetterich et al. (2012) investigated the WTP of German consumers for sustainability features in a car's interior by direct inquiry but used haphazard sampling. They found a WTP for a price premium of 3.5% for a medium-sized vehicle (Hetterich et al. 2012).

2.3.2 Choice of WTP method

Figure 1 shows an overview of the methods that can be used to determine WTP of consumers and how they can be classified. For the study in this research, the group of individual measurements was chosen as the identification of a possible target group for sustainability features out of a large group of individual customers should be possible. As lotteries constitute non-realistic purchase situations—for instance, it is unsure for participants whether they receive the intended product—this method might elicit unrealistic and more strategic purchasing behavior from participants (Voeth and Niederauer 2008). Auctions are more appropriate to assess WTP for scarce products, like rarities (Backhaus et al. 2005). Furthermore, auctions could lead to a distorted purchasing behavior as strategic and competitive motivations as well as the anticipated excitement of a win can influence the WTP of participants (Kagel 1995; Ding et al. 2005). Preference data can be gathered by direct inquiry (e.g., Van-Westendorp analysis) or indirect inquiry (e.g., Conjoint Analysis). Sattler and Nitschke (2003) showed that direct inquiries can produce results closer to the real WTP than conjoint analysis. But Backhaus et al. (2005) found that the same study design can yield the opposite results with conjoint analysis landing closer to the real WTP. Therefore, it was decided for this study to combine both methods. Conjoint analysis is considered to be replicating a realistic purchase situation and is widely accepted and applied in market research (Baier and Brusch 2009).

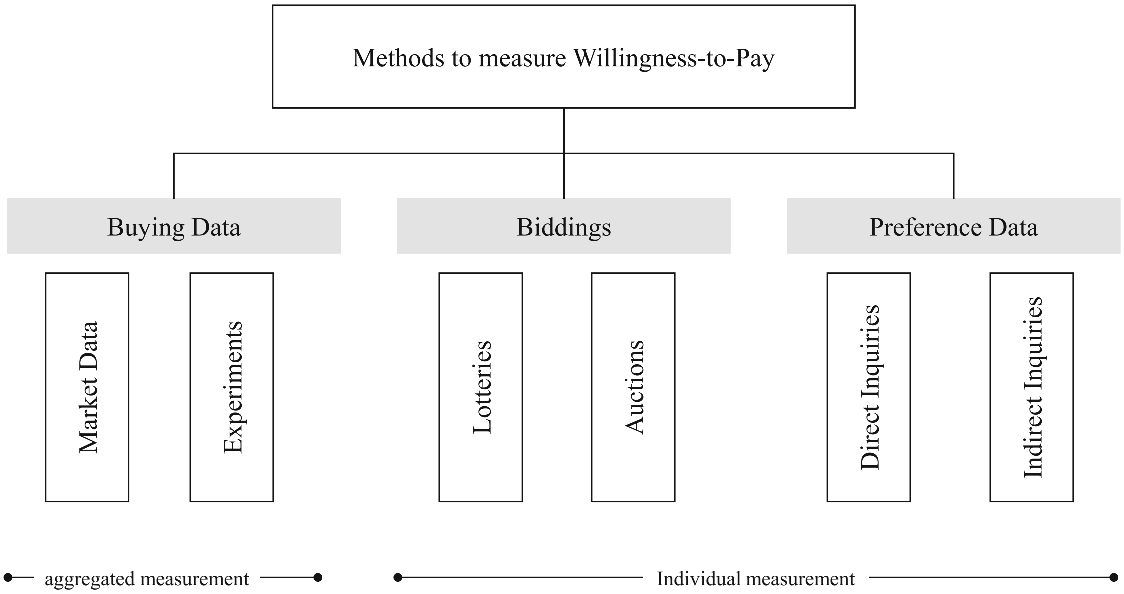


Fig. 1 Methods for measuring willingness-to-pay—adapted from Skiera and Gensler (2002), Sattler and Nitschke (2003), and Voeth and Niederauer (2008)

The primary objective of the Van-Westendorp analysis is to validate and/or extend the results of other methods like conjoint analysis (Simon and Fassnacht 2009). The extension of the study with a Van-Westendorp inquiry was considered to be a reasonable increase in volume of the study and effort for participants. The choice-based conjoint analysis (CBCA) is considered to be the best variation of conjoint analyses in regards to the quality of prognosis (Kaltenborn et al. 2013).

2.3.3 Choice-based conjoint analysis

The main goal of the CBCA is to capture the selection process of respondents as realistically as possible. In this method, not more than five to seven product properties should be tested as participants have difficulties evaluating too many alternatives (Kaltenborn et al. 2013). As opposed to ranking of product alternatives in the classical conjoint analysis, the CBCA asks respondents for a discrete choice of the best alternative out of varying sets of three to five alternatives (Baier and Brusch 2009; Kaltenborn et al. 2013). The assumption is that respondents will always choose the alternative with the highest overall benefit for them (Ben-Akiva and Boccara 1995). One of the choice options within a set is always a no-choice option if a respondent would choose neither of the given alternatives. In this way, the purchase decision of consumers is more realistically captured, which in other methods like classical conjoint would require modifications like a limit card (Cohen 1997; Baier and Brusch 2009). The number of choice tasks, i.e. choosing the best alternative out of a set, should not exceed 20 (Johnson and Orme 1996). Even though the CBCA in its roots is an aggregated measurement method (Kaltenborn et al. 2013), the application of the hierarchical Bayesian method (HB) enables the estimation of individual utility values (Baumgartner and Steiner 2009). Utility values represent the utility or strength of preference (positive values) or the strength of rejection (negative values) of a product attribute specification for a potential customer. They have to be considered relative to the other evaluated attribute specifications rather than as an absolute indication of an independent preference value.

The choice of product attributes and their specifications are crucial for the results of the analysis. Backhaus et al. (2011a) list several criteria that should be considered:

- Relevance: attributes should have relevance for purchase decision
- Controllability: the manufacturer should be able to influence the attribute
- Independence from preference: utility of an attribute should be independent from other attributes
- Realistic: attributes should be realizable
- Compensatory: one attribute can compensate for another
- No knockout (KO) criterion: an attribute or a specification of it should not lead to immediate rejection of the product

2.3.4 Advanced-Van-Westendorp analysis

Consumers do not have a fixed WTP or "definitive" acceptable price but rather an acceptable price range (van

Westendorp 1976). The Van-Westendorp analysis is trying to identify this acceptable price range and additionally an optimal as well as an “indifference” price for consumer. The indifference price can be interpreted as average market price or median (Müller 2008). To determine these price levels, respondents have to answer four questions (van Westendorp 1976; Reinecke et al. 2009):

1. Cheap: Below what price do you consider the product as cheap/a good deal?
2. Expensive: At what price would you consider the product as expensive but would still consider buying it?
3. Too expensive: At what price would you consider the product as too expensive and would not consider buying it?
4. Too cheap: Below what price would you consider the product as too cheap and therefore doubt its quality?

The cumulative evaluation of those four price levels and the share of respective respondents result in four price curves (van Westendorp 1976; Müller 2008). From the intersection of those curves, four price levels can be identified. First, the indifference price point (IPP) which could be interpreted as market average price. Second, the optimal price point (OPP) which denotes the optimal price for the consumer. The point of marginal cheapness (PMC) and point of marginal expensiveness (PME) denote the lowest and highest price of the acceptable price range, respectively. Especially when evaluating new products, the Van-Westendorp analysis is a good choice (Reinecke et al. 2009). As with all methods that directly inquire the WTP of consumers, the over-estimation of the importance and thus the amount of the price premium is a drawback of the Van-Westendorp analysis (Reinecke et al. 2009). List and Gallet (2001) and Murphy et al. (2005) found that hypothetical WTP often differs by factors of 1.5 to 3 from real WTP.

Roll et al. (2012) extended the Van-Westendorp analysis by translating the consumer relevant price levels into a demand function by which users of the method can deduct at which price level the highest revenue is to be expected. Thus, the method helps to shift from a consumer’s to a company’s perspective and can determine the optimal price for the product from a company’s viewpoint.

2.4 Increasing the usability of LCSA results to support decision making

The LCSA framework as defined by Klöpffer (2003, 2008) and Finkbeiner et al. (2008) is operationalized at an automotive company and therefore applied to an entire vehicle. The implementation of LCSA at an automotive company has to overcome several challenges (Tarne et al. 2017). In the following, the main challenges and how they are overcome are presented and an approach is introduced to putting the LCSA framework into practice at an automotive company. The consistent execution of the three life cycle-based analyses is a point to be resolved, especially as S-LCA is comparatively low in method maturity (Finkbeiner et al. 2010; Martínez-Blanco et al. 2014; Klöpffer and Grahl 2014; Karlewski 2016). One initiative to present practical guidance on how to assess the social impacts of products has been presented by the Roundtable for Product Social Metrics that produced the Handbook for Product Social Impact Assessment (Fontes et al. 2015; Fontes et al. 2016). The sensible integration of LCC alongside LCA and S-LCA is also in debate. The point has been made by Jørgensen et al. (2010), Wood and Hertwich (2013), and Neugebauer et al. (2016) that LCC in its current form is unfit to complement LCSA and S-LCA in a meaningful manner. Especially, the focus of the analysis and the inherent conflict of goals (minimize costs for consumers but maximize value added for society) was called out by Wood and Hertwich (2013). As the focus of this research lies in putting LCSA into practice at an automotive company, the ideal decision support from a company’s viewpoint should be ensured first. Thus, the applied framework is geared towards enabling decision making at an automotive company and the company perspective is chosen for the LCC performance. This way, the conflicting targets of LCC are resolved as the target is interpreted from the company’s point of view. There is also no consensus on how to effectively support decision making, especially when trade-offs within or between LCSA dimensions occur (Traverso et al. 2012; Tarne et al. 2017). Furthermore, the indicators for each dimension are chosen in line with the company’s (product) sustainability strategy and targets which makes it easier to interpret whether an impact is positive or not, i.e., aligned with the strategy/targets or not. Another challenge to put LCSA into practice at an automotive company is the lack of monetary interpretation of product sustainability impacts (Tarne et al. 2017).

In the vehicle development process, components are often used as the smallest manageable unit supported by the trend of increasingly outsourcing component development to suppliers (Ciravegna et al. 2013). Therefore, the “resolution” at which the vehicle is assessed is the component level. To identify the greatest potentials to improve product sustainability impacts at the component level, the focus is, in a first step, laid at the production phase. The “initial set” in Fig. 2 shows the theoretical assessment of a vehicle’s components regarding their LCSA performance in the production phase. The proposed way of applying LCSA to a vehicle introduces criticality points to measure the overall (negative) impact of the respective component on the concerned sustainability dimension. The criticality points of the approach shown in Fig. 2 are determined by the relative performance of the respective component in the respective sustainability dimension compared to the other components that make up a vehicle. The criticality

Initial set:

	35%	34%	31%	
Component	**Criticality Points LCA**	**Criticality Points LCC**	**Criticality Points S-LCA**	**Criticality Points Overall LCSA**
A	72	11	100	60
B	82	17	42	48
C	20	52	45	39
D	…	…	…	…

Improvement measures for component B:

	35%	34%	31%	
Component	**Criticality Points LCA**	**Criticality Points LCC**	**Criticality Points S-LCA**	**Criticality Points Overall LCSA**
A	72	11	100	60
C	20	52	45	39
B*	44 ⇩	38 ⇧	25 ⇩	36
D	…	…	…	…

?

Fig. 2 Evaluation of an improvement measure for component B. The impacts on the environmental (LCA) and social (S-LCA) dimension have improved from the company's perspective while the impact on the economic (LCC) dimension has deteriorated. Even though applying equal weights, the overall change in criticality would be acceptable (the overall LCSA criticality points of B* are lower than those of B), the increase in costs, however, would most probably constitute a deal-breaker in a business context. Thus, the decision for improving component B is uncertain. The figure shows fictional values for demonstration purposes

points are determined based on any selection of indicators deemed relevant to the company's product sustainability goals. For example, the LCA dimension of the evaluation scheme in Fig. 2 could be determined based on the greenhouse gas (GHG) emissions and the water consumption during the production of the respective component. That means the amount of GHG emissions and water consumption are compared between components. Outranking methods like VIKOR (Opricovic and Tzeng 2004), PROMETHEE (Brans 1982; Behzadian et al. 2010), or ELECTRE (Benayoun et al. 1966; Govindan and Jepsen 2016) would be suitable for this comparison, because they transform absolute metrics from any measurement unit (e.g., kg, m^3, € etc.) into relative metrics that are dimensionless. That means that they would transform the impacts of GHG emissions and water consumption into dimensionless scores that can be normalized to a scale of 0–100 and be aggregated. The proposed approach awards the component with the highest combined impact in an LCSA dimension 100 criticality points in the respective dimension. The component with the lowest impact is assigned zero criticality points. For instance, component A in Fig. 2 has the highest overall social impact (e.g., due to the high share of its materials coming from socially critical sources) reflected in the 100 criticality points in the S-LCA column. The environmental impacts are also comparatively high with 72 criticality points in the LCA dimension, meaning that the combined GHG emissions and water use during its production amount to 72% of the component that performs worst in the LCA dimension. As the relative evaluation of this approach puts the absolute values of all considered components into relation to the span between the top and worst performing component (Opricovic and Tzeng 2004; Opricovic and Tzeng 2007), the awarded score will change as soon as the best or worst performing component is changed.

In principle, the concept of strong or weak sustainability can be translated into this evaluation. Following the concept of strong sustainability, no compensation between sustainability dimensions is allowed. Therefore, the performance in the LCA dimension would be the priority. Deterioration in this dimension would not be accepted, and only if this dimension is not negatively impacted, the impact on the S-LCA dimension would be considered. And finally, if LCA and S-LCA are not negatively impacted, the LCC dimension could be considered. As mentioned in the introduction, this concept would be preferable when the “Limits to Growth” as postulated by Meadows et al. (1972, 2007b) are to be considered.

However, as this operationalized framework tries to garner more support within the industry, it follows the weak concept of sustainability (Davies 2013) and therefore compensations or trade-offs between dimensions are possible. To adopt this interpretation at a company could entail the risk that it is accused of green-washing (Parguel et al. 2011; Kruschke and Vanpaemel 2015) as it would accept deterioration in environmental impacts for the benefit of increased profits (Figge 2005).

For the presentation of the mechanics of the operationalized LCSA approach in the following, equal weights are assumed. The overall LCSA criticality points can be used to:

1. Determine the component with the highest overall impact on product sustainability
2. Evaluate improvement measures

The first application of the operationalized LCSA method is clearly visible in the ranking itself when components are sorted by the overall LCSA criticality points. The component with the highest overall LCSA criticality points represents the one with the highest overall impact on product sustainability. The list could therefore help practitioners in prioritizing their search for product improvement measures. The second application is visualized in Fig. 2.

At the top of Fig. 2, the evaluation of the initial set of components is shown. A possible improvement measure in the production of component B, e.g., the switch from grid mix electricity to locally generated renewable electricity in the production process, changes the impacts on the three sustainability dimensions. The environmental and social impacts have improved, e.g., due to less environmental impacts and local employment, but the economic impact has declined, e.g., due to higher electricity prices.

One crucial step of this operationalized framework is the aggregation of indicators within and the criticality points between three separate dimensions to an overall sum of LCSA criticality points in order to support decision makers in their assessment. Depending on which aggregation logic is applied, these changes in criticality points could result in a change of the overall LCSA criticality points giving decision makers the clear indication that the evaluated improvement measure would contribute to the overall improvement of product sustainability. This challenge is addressed by Tarne et al. (2018) through the introduction of weights to the sustainability dimensions. However, in a business context, an increase in costs would most likely lead to the rejection of the improvement measure for component B. Here, the concept of monetization, and the Product Sustainability Budget in particular, offers a solution.

The PSB is not a way to monetize sustainability impacts in the sense of determining the causal relationship between impacts caused along a product’s life cycle and potential costs to society or a company. The PSB rather gives an interpretation from a business point of view how much product sustainability contributes to additional revenue and how much of this can be invested in product sustainability improvement measures.

3 Methods

To determine the willingness-to-pay for sustainability features, a choice-based conjoint analysis in combination with an Advanced-Van-Westendorp analysis was carried out (Sect. 3.1). The creation of the Product Sustainability Budget drawing from the WTP study results is presented in Sect. 3.2.

3.1 Determining the customer value of a sustainability package in a premium car

As laid out in Sect. 2.2, the focus of the study was placed on the customer relevant aspects of sustainability within a premium car. Therefore, a fictitious sustainability interior package was devised that was characterized by several measures that could be perceived by customers and had a positive impact on at least one environmental category (e.g., CO_2 emissions or resource use). To reduce social bias regarding sustainability and to simulate a more realistic purchase decision, the interior sustainability package was presented along four other vehicle properties (price, drive train, engine power, and consumption) to implicitly derive the utility of the said package for potential consumers.

3.1.1 Selection of stimuli

To avoid the number of level effect, i.e., the over-estimation of attributes with more specifications (Baier and Brusch 2009), and to enable a symmetric study design, the same number of specifications per attribute were set. To also avoid cognitive overload of respondents, the number of attributes were limited at five as the range of the maximum suggested number of attributes is five to seven (Sichtmann and Stingel 2007; Kaltenborn et al. 2013). Table 1 shows the chosen attributes and specifications.

The attribute “Price” was chosen as it was deemed an important factor in the purchase decision. The specifications were chosen for a lowly motorized and equipped middle class limousine (low), a medium motorized equipped middle class limousine (medium), and a highly motorized and equipped middle class limousine (high). As it is an important attribute of vehicles and at the same time ties into the sustainability properties, the “Drive Train” was also selected as an attribute.

Table 1 Chosen attributes and attribute specifications for the CBCA

Attribute	Attribute specification		
Price	30,000 €	35,000 €	40,000 €
Drive train	Combustion	Hybrid	Electric
Engine power	116 HP	150 HP	190 HP
Interior package	Standard	Luxury	Sustainability
Consumption per 100 km	Low (3.8 l/9 kWh)	Medium (5.5 l/13 kWh)	High (7.2 l/18 kWh)

HP horsepower

Combustion hereby included gasoline and Diesel powered vehicles, hybrid denoted the combination of combustion engine and externally rechargeable battery-electric engines. Electric referred to a purely battery-electric vehicle. As the envisioned target audience was of the premium segment, “Engine Power” was also chosen as an attribute. The specifications were based on the mostly used motorizations of the chosen middle class limousine. As the preference of sustainability aspects should be measured indirectly but be customer relevant, a sustainability interior package was defined. A dashboard from natural fibers, trim strips of FSC certified wood and a cup-holder of recycled ocean plastic characterized the fictitious sustainability package. Alternatives for the attribute “Interior Package” were a standard package (dashboard of plastics, trim strips of painted plastic, and a cup holder of plastic) and a luxury package (dashboard covered with leather, trim strips of coated exotic wood or metals like aluminum or chromium, and a cup holder of carbon). As fifth attribute, “Consumption” was chosen as it constituted an economic (fuel costs) and environmental (CO_2 and other emissions) factor in a car purchase. Respondents could select the specifications “low”, “medium”, and “high”. To give an indication for combustion, hybrid, and electric vehicles, the levels of consumption were given in liters (l) and kilowatt-hours (kWh) per 100 km (c.f. Table 1). Attributes like brand, vehicle type (e.g., sports utility vehicle (SUV), sports car, station wagon), or color were deliberately not chosen as personal taste should be excluded as best as possible from the conjoint analysis. To nevertheless respect these relevant factors in the virtual purchase situation, respondents were informed at the beginning of their choice tasks that they should envision already having chosen a type, brand, and color for their car and that they would be asked to choose additional features in the following steps. For price, engine power, and consumption, the vector model was assumed for the utility values whereas drive train and interior package were assumed to follow no particular function, thus the part-worth model.

With three specifications per attribute, 243 (3^5) stimuli were possible. To reduce the number of possible alternatives to a manageable amount, a representative set of 20 stimuli was derived via study design algorithms from XLSTAT (=reduced design). In the choice task, respondents were asked to pick their favorite alternative out of three possible stimuli or select the “no choice” option. Respondents were asked to complete 20 of those choice tasks. The stimuli were additionally represented by visual symbols in order to make them easily understandable and discernable.

3.1.2 Measurement of WTP

The Advanced-Van-Westendorp analysis consisted of five questions. The first question was designed to determine whether respondents would consider buying the sustainability package. The sustainability package was again presented to respondents who had to answer the following question with yes or no: “Would you generally consider buying this sustainability package with your next car?” Only respondents who answered “yes” were included in the analysis to derive the ideal price at which the sustainability package should be sold. The question was followed by the four questions regarding the four price levels cheap, too cheap, expensive, and too expensive (c.f. Sect. 2.4.4).

3.1.3 Sample selection

A sample size of $n = 250$ completed surveys for the German market was completed via an online panel service that was certified according to ISO 26362. To identify potential customers of interest, respondents should be car holders that were involved in the decision of the purchase of a car, prefer buying new cars (where the choice of an interior package is relevant and can be influenced), and be interested in buying a car in the future. As the premium car sector was of interest, respondents were asked which brands they would consider in their next car purchase. If no premium brand was present, they were excluded from the survey. The sample selection was ensured by respective screening questions at the beginning of the survey that halted the survey if the criteria were not met.

3.1.4 Study execution

At the beginning of the questionnaire, respondents were asked several screening questions to ensure that they were potential customers of interest (c.f. Sect. 3.1.3). Afterwards, they were introduced to the three different interior packages. Table 2 shows how they were characterized. In addition, respondents

Table 2 Characterization of the different interior packages

	Standard	Luxury	Sustainability
Dashboard cover	Plastic	Leather	Natural fiber
Trim strips	Plastic (varnished)	Exotic wood	FSC certified wood
Cup holder	Plastic	Carbon fiber	Ocean plastic

were shown an exemplary picture of the respective interior package, where the differences in the materials could be seen.

Respondents were then presented with 20 choice tasks out of which they were asked to select their preferred option. Figure 3 shows an exemplary choice task.

After completion of this task, they were shown the sustainability interior package again and asked the four Advanced-Van-Westendorp questions (c.f. Sect. 2.3.4). As a last step, they were asked about socio-economic status, i.e., monthly income, age, occupation.

3.1.5 Evaluation of study

The hierarchical Bayesian method was used to iteratively estimate the individual utility values based on the overall distribution within the sample. In the following, the method is described in short. For more information on the hierarchical Bayesian method, the reader is referred to Gelman et al. (2004). First, the normal distribution was assumed as underlying distribution of the individual utility values. The choice pattern of respondents was characterized by applying a multinomial logit approach. In a following iterative approach, the individual utility values were estimated by considering individual respondents choice pattern and the underlying value distribution of the aggregated sample. As pointed out in Sect. 2.3.3, the resulting utility values—as in all conjoint analyses—are relative values, i.e., they can only be interpreted in the present approach and only in relation to each other. The built-in hierarchical Bayesian method (HB) algorithm in XLSTAT was used for the estimation of utility values. The following analyses were carried out via SPSS.

To identify the target group for the sustainability package, i.e., the fraction of the study sample that would choose the sustainability package to be in their new car, the respondents were filtered by the following criteria:

- Respondents who preferred the sustainability package to the other interior packages, i.e., the utility value of the sustainability package was higher than for the other packages
- The preference for the sustainability package should have non-negligible effect on the decision for the respective stimulus, i.e., utility values > 1
- Respondents who answered "yes" to the question whether they would consider buying the sustainability package in the Advanced-Van-Westendorp analysis

3.2 Creation of the product sustainability budget and link to LCSA framework

This section lays out a proposition to translate the WTP for customer relevant sustainability features (as measured by the study in Sect. 3.1) to the entire product in order to enable the implementation of measures that improve product sustainability but have no direct customer benefit. Figure 4 shows the mechanism of the proposed Product Sustainability Budget.

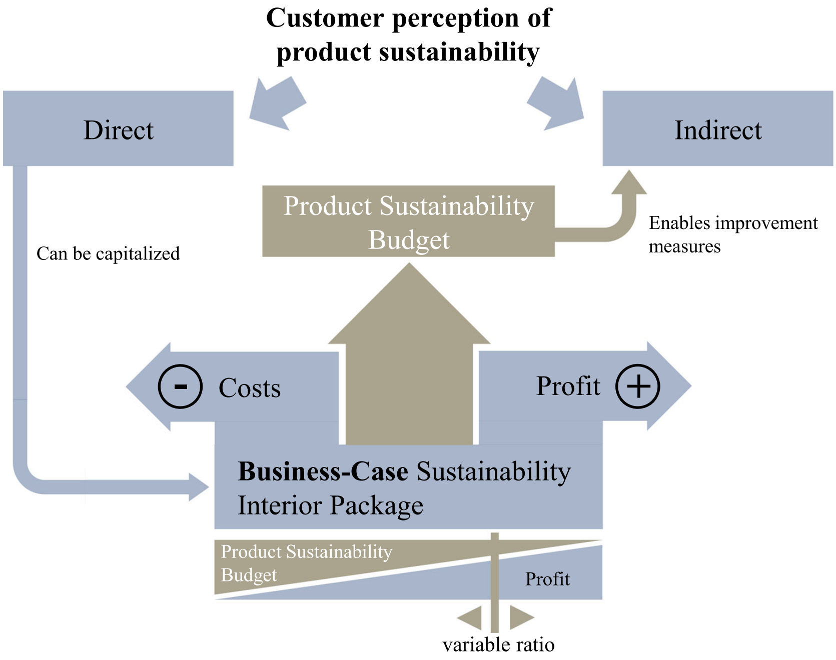
The premise for the Product Sustainability Budget starts at the customer perception of sustainability. It is divided into directly perceivable aspects (like a sustainable interior) and indirectly perceivable aspects (like an improved carbon footprint). Directly perceivable product aspects can be capitalized in the

Please select your preferred option.

Product A		**Product B**		**Product C**		
Price:	30,000 €	Price:	30,000 €	Price:	35,000 €	None of the presented alternatives
Drive train:	Hybrid	Drive train:	Electric	Drive train:	Combustion	
Engine Power:	190 HP	Engine Power:	116 HP	Engine Power:	150 HP	
Interior Package:	Standard	Interior Package:	Sustainability	Interior Package:	Sustainability	
Consumption:	Low	Consumption:	High	Consumption:	Low	
○		○		○		○

Fig. 3 Exemplary choice task that respondents of the CBCA were presented with

Fig. 4 Mechanism of the product sustainability budget

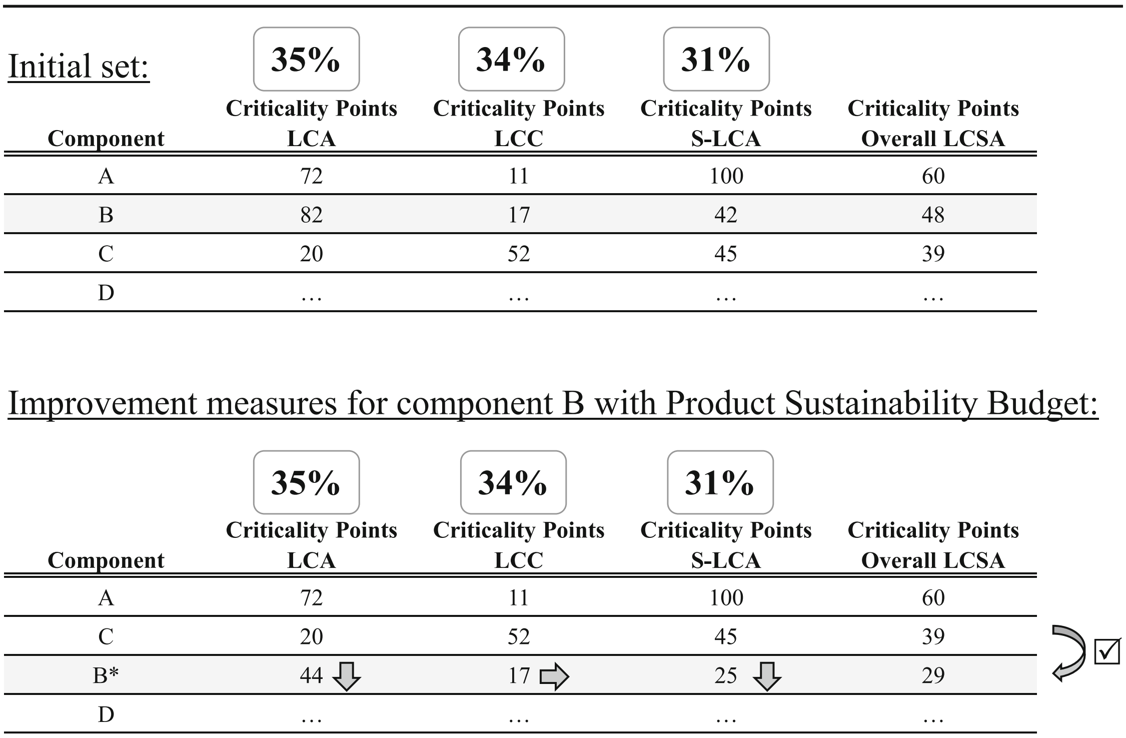


market as the WTP study in this paper was designed to evaluate. Indirectly perceivable aspects cannot, or at least hardly, be capitalized in the market but nevertheless contribute to a more sustainable product. The process depiction in Fig. 4 is obviously simplified to demonstrate the general mechanism. Every business would apply its own approach to business case calculation here. The revenues incurred by the sale of, for instance, a sustainable interior package would have to cover the costs for production, distribution, administration etc. of this package. If the business case were positive, the rest would go as profit towards the company's balance sheets. The proposition in this paper is to declare a share of this profit as a Product Sustainability Budget. That means, this money would not go towards the company's balance but would rather be re-invested into indirectly perceivable product sustainability features.

This diverted profit could be connected with the operationalized LCSA framework (c.f. Sect. 2.4) to support decision making based on LCSA results. That means that the LCSA results as determined based on the operationalized framework were used to identify a set of improvement measures to be undertaken. On this set of improvement measures, the PSB was applied to support the decision-making process at an automotive company under the conditions of costs constraints. The study setup in this paper was done independently from the considered LCSA dimensions in the operationalized framework. This poses no problem as the Product Sustainability Budget basically provides an amount of money with which sustainability improvement measures can be financed. The way this amount is determined is based on the determined customer added value and a business case logic (as shown in Fig. 4). In this way, product sustainability could be improved even if proposed improvement measures or product features would pose a trade-off situation in the LCSA framework due to cost increase. For example, if 50 € of the profits from the sale of a sustainability interior package should go towards the Product Sustainability Budget and 20% of customers choose this interior for their car, every vehicle can be allocated an allowance from this budget of 10 €. If we look at the improvement measure of component B (c.f. Fig. 2) again and the calculated price increase per unit due to this measure would amount to 5 €, the evaluation would look as displayed in Fig. 6.

In Fig. 2, the improvement measure for component B posed an inacceptable (at least in a business context) trade-off due to increased costs. Once the Product Sustainability Budget is introduced, the cost increase can be covered and thus does not affect the performance in the LCC dimension. The LCC criticality points for component B* remain unchanged in Fig. 5 enabling a clear decision support for the improvement measure.

The proposition of this paper is to enhance the decision-making process for product sustainability improvement measures that are evaluated by LCSA at an automotive company. The LCSA framework to be augmented by the PSB has been made more operational by previous steps—especially the introduction of weights (Tarne et al. 2018). These weights reflected the weightings of decision makers at an automotive company and were used to determine an overall LCSA impact



Initial set:

Component	35% Criticality Points LCA	34% Criticality Points LCC	31% Criticality Points S-LCA	Criticality Points Overall LCSA
A	72	11	100	60
B	82	17	42	48
C	20	52	45	39
D	…	…	…	…

Improvement measures for component B with Product Sustainability Budget:

Component	35% Criticality Points LCA	34% Criticality Points LCC	31% Criticality Points S-LCA	Criticality Points Overall LCSA
A	72	11	100	60
C	20	52	45	39
B*	44 ⇩	17 ⇨	25 ⇩	29
D	…	…	…	…

Fig. 5 Linking of the Product Sustainability Budget to the operationalized LCSA framework enables the clear decision support in trade-off situations where costs would rise. The figure shows fictional values for demonstration purposes

of a car or its components. That means that trade-offs between sustainability dimensions were deemed acceptable, thus putting the approach into the realm of weak sustainability. However, human rights violations were explicitly excluded from the trade-off evaluations as they were deemed as KO criteria from the company's point of view.

How the Product Sustainability Budget provides a solution space for finding the ideal set of measures for overall product sustainability improvement is shown in Sect. 4.2.

4 Results and discussion

In this section, the results of the study on the customer value of a sustainability package in a premium car are presented (Sect. 4.1). In Sect. 4.2, the way of exploiting the solution space provided by the Product Sustainability Budget when linked to the operationalized LCSA framework is laid out.

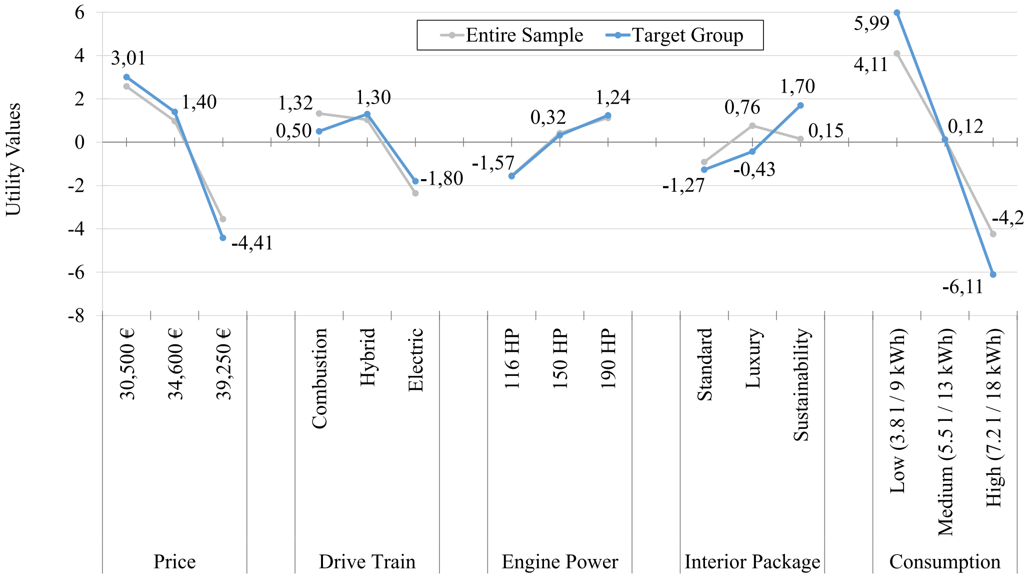
4.1 Customer value of a sustainability package in a premium car

Out of the sample of 250 respondents, 30 had to be excluded as their share of "no-choice" answers in the 20 choice tasks was too high to be used in the analysis. The utility values of the sustainability package are presented in Sect. 4.1.1 for the entire sample and for the target group of the sustainability package. The price at which this sustainability package should be sold is presented in Sect. 4.1.2.

4.1.1 Utility values of the sustainability interior package

Figure 6 shows the utility values of the investigated attributes and their specifications for the entire sample and the target group for the sustainability package (the identification process of the target group is described in Sect. 3.1.3). The target group for the sustainability package constituted 19% of the entire sample (42 respondents out of an entire sample of 220 valid responses).

The utility values in Fig. 6 indicate that the sustainability package had a clearly higher utility value (1.70) for the target group than for the entire sample (0.15). This should be expected as the members of the target group were defined to be respondents who assigned the sustainability package a utility value of more than 1.0. This was also reflected by the higher overall importance of the attribute "Interior Package" for this group, which influenced their purchase decision with 14% compared to 11% for the entire sample. The entire sample preferred the luxury package (utility value = 0.76) to the other packages which was expected for customers from the



Overall Importance					
Entire Sample	23.4%	23.7%	15.8%	11.1%	26.1%
Target Group	23.8%	17.3%	12.1%	13.5%	33.3%

Fig. 6 Utility values of the investigated attributes and their specifications for the entire sample and the target group for the sustainability package (top). The overall importance of the respective attribute in the purchase decision for the entire sample and the target group is shown at the bottom

premium car segment. But even for the entire sample, the sustainability package was clearly preferred to the standard package (utility value of 0.15 vs. −0.91). Both groups did not prefer the standard package.

The determination of the utility values for consumers of the respective attributes is estimated by relying on utility value models that describe the utility function of an attribute. The most common are the vector, the ideal point, and the part-worth model (Backhaus et al. 2011b). The vector model assumes a linear relationship between attribute specification and utility value (e.g., price; the cheaper the better). The ideal point model assumes an optimal attribute specification below or above which the utility value decreases (e.g., sweetness coffee; too much or too little is worse). The part-worth model does not assume any relationship between attribute specification and utility value. Especially for discrete attribute specifications, like color, this model is applied.

For the attributes price, engine power, and consumption, the utility values followed the vector model, as assumed. The higher the price was, the more negative the utility value. The target group did not set itself clearly apart in its utility function regarding price from the entire sample hinting at a consistent evaluation of price within the premium car segment. This was supported by the medium price (35,000 €) still having a decent positive utility value (1.40) meaning that the entire sample had a high price tolerance. Engine power also followed a “the higher, the better” logic which was also not surprising for the premium car segment. Here, target group and entire sample also showed consistent evaluation. When looking at consumption, the evaluation of the target group differed slightly from the entire sample. While both consistently deemed higher consumption less favorable, the target group placed more importance on this attribute. This was reflected by the higher utility values and the overall importance of this attribute (33% vs. 26% of the entire sample). As price was consistently valued between both groups, the economic importance of consumption should also be expected to be the same. Thus, the difference in importance between both groups could be due to the environmental interpretation of this attribute by the target group, meaning that people who preferred the sustainability package also wanted their car to consume less fuel, be it conventional or electricity, to reduce their environmental impact further.

Both groups preferred hybrid and combustion to the electric drive train. This might have been due to skepticism towards the range and infrastructure of this alternative as the hybrid drive train was clearly favored to the combustion engine by the target group. The entire sample still marginally favored the combustion engine which also posed no surprise as the sample, like the premium car segment, was mostly made up of consumers of advanced age (51% of respondents were older than 45 years) indicating a certain reliance on familiar technologies. However, the hybrid drive train was an acceptable alternative in the premium car segment with the utility values of both groups ≥ 1.03.

Figure 7 shows the range of the utility values for the entire sample.

Interestingly, the attributes with higher weight in the purchase choice (e.g., price—c.f. Fig. 6) exhibited larger ranges of utility values than attributes with less weight, e.g., the interior package. This indicated a lower consistency in the valuation of the more important attributes throughout the entire sample. The highest consistency in utility values across the sample was recorded for the sustainability interior package, the medium consumption, the medium engine power, the medium price, and the standard interior package. This means that the conclusions drawn for the utility of a sustainability interior package derived from this research can be deemed robust, as the premium car segment appeared to have a consistent evaluation of the utility of this sustainability package.

4.1.2 WTP for a sustainability package

Out of 250 respondents, 179 answered “yes” to the question if they would generally consider buying the sustainability package. In Fig. 8, the results of the Advanced-Van-Westendorp analysis for the target group of the sustainability package are presented as they represent the relevant group of customers for determining the price at which to offer such an interior package.

In Fig. 8, the results of the standard Van-Westendorp and Advanced-Van-Westendorp analysis are shown together. The four price points were retrieved by the standard Van-Westendorp analysis. The acceptable price range for the sustainability package for the target group ranged from 0.5% (i.e., the point of marginal cheapness, PMC) of the price of a middle class limousine to 1.7% (i.e., the point of marginal expensiveness, PME). The optimal price point (OPP) and indifference price point (IPP) laid at 1.0%. The price for highest relative revenue and thus the optimal price from a company’s viewpoint laid at 1.3% and 1.7%. This was half of what Hetterich et al. (2012) found as customer’s willingness to pay a price premium for sustainable interior components. However, the studies are not readily comparable as the study of Hetterich et al. used haphazard sampling and determined WTP by pre-defined Likert scale points.

When social bias for a direct inquiry of WTP is considered (c.f. Sect. 2.4.4), the actual WTP would lie 1.5–3 times lower (List and Gallet 2001; Murphy et al. 2005). Thus, a price of as low as 0.4% of the price of a middle class limousine (=worst case: 1.3%/3) or as high as 1.1% (=best case: 1.7%/1.5) should be aimed for.

4.2 Sustainability budget and link to LCSA framework

The values in this section are fictional values. They are used for illustrative purposes to show how the Product Sustainability Budget, derived from a business case calculation, could be used

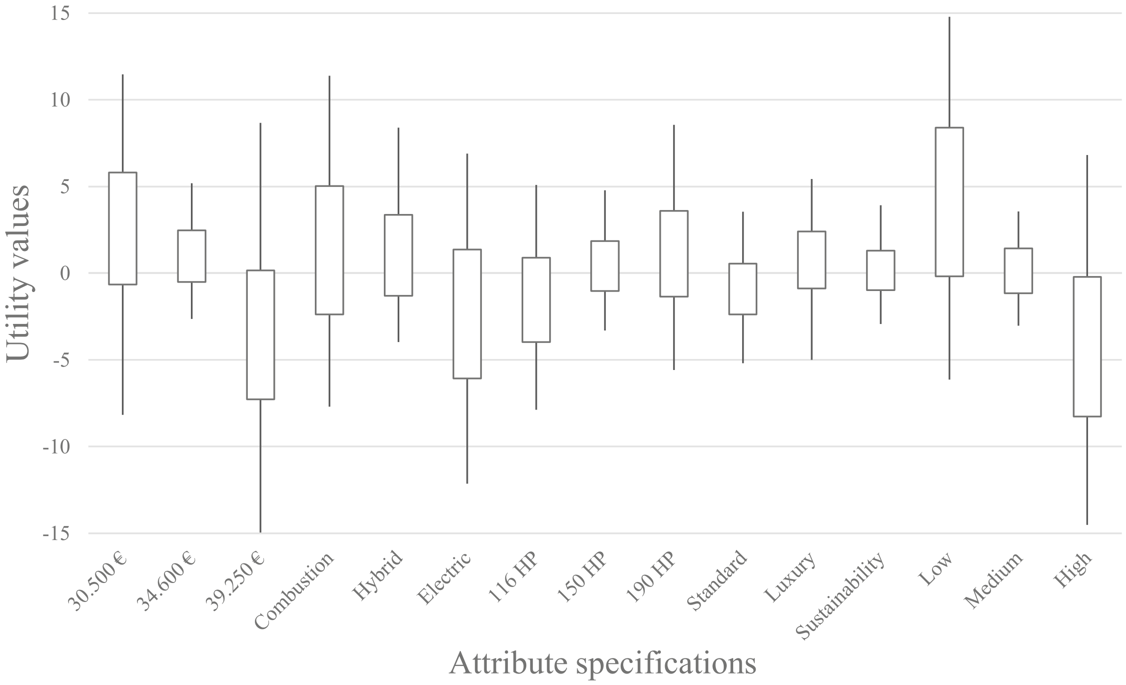
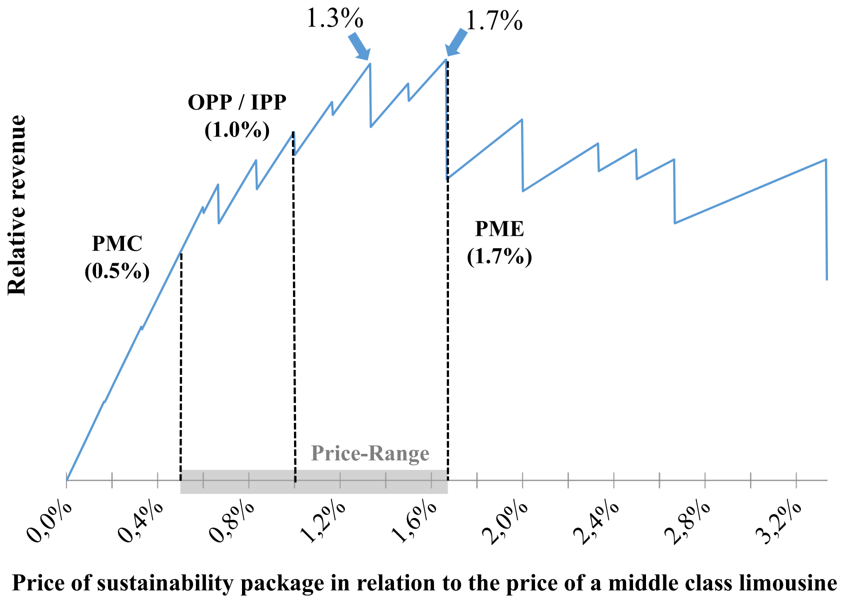


Fig. 7 Range of utility values for the entire sample. Top and bottom of the box-plot indicate the minimum and maximum. The box indicates the standard deviation around the average utility value

Fig. 8 Results of Advanced-Van-Westendorp analysis for the target group of the sustainability package



to enable decision support within the LCSA framework if costs would increase. For this research, it was assumed that the business case of Sect. 3.2 in combination with the results in Sect. 4.1 yielded a Product Sustainability Budget of 10 €. That means that for every product, up to 10 € of cost increase could be compensated as they have already been earned by the sustainability interior package. The Product Sustainability Budget therefore opened up a solution space in which the best solution for overall product sustainability improvement according to the operationalized LCSA framework could be sought. If several improvement measures are available that differently improve LCA or S-LCA performance while differently increasing costs, the optimal overall solution might not be clearly visible. Table 3 lists several theoretical measures to improve product sustainability impacts.

To find the ideal solution, linear optimization was applied using Excel Solver. The Product Sustainability Budget was taken as a restraining factor while the sum of LCA and S-LCA criticality points was optimized. This yielded that the combination of improvement measure II and VI would yield the highest improvement in product sustainability (− 66 criticality points) while still staying within the limit of the Product Sustainability Budget (9.40 €). That means that it allowed the selection of the ideal improvement measure from several alternatives from a company's perspective.

Even though the impacts themselves have not been subject to monetary valuation in this research, the Product Sustainability Budget enabled the decision support in trade-off situations with increased costs by translating customer added value to the LCSA framework evaluation.

The introduction of weights by Tarne et al. (2018) enabled the consideration of trade-offs and adopted the weak interpretation of sustainability. Therefore, crucial conditions of the interpretations of the trade-offs between sustainability dimensions have been fixed already. Nevertheless, as the two concepts—the LCSA framework and the PSB—are independent from each other, the evaluation logic of the LCSA framework does not influence the way that the PSB can be applied.

The WTP study to derive the current PSB only considered the environmental dimension. This might be confusing for readers as the PSB was then used to support decision making

Table 3 Theoretical measures to improve product sustainability impacts

Improvement measure	Costs	Criticality points LCA	Criticality points S-LCA	Total	Criticality points/€
I	6.70 €	− 30	− 17	− 47	− 7.0
II	3.40 €	− 10	− 12	− 22	− 6.5
III	4.20 €	− 15	− 15	− 30	− 7.1
IV	5.00 €	− 25	− 10	− 35	− 7.0
V	2.10 €	− 9	− 4	− 13	− 6.2
VI	6.00 €	− 25	− 19	− 44	− 7.3

at a company based on LCSA results where all three dimensions were evaluated. It is therefore reiterated that both concepts evaluated two different aspects of product sustainability. The LCSA framework evaluated the product sustainability impacts while the PSB evaluated the customer added value of product sustainability. The PSB was used to give financial leeway for a company in its decision process. The limited evaluation of just environmental impacts in a sustainability interior package therefore might have led to an underrepresented WTP by potential customers and therefore to a lower PSB. However, the focus on this one sustainability dimension does not render the results or the PSB invalid for use with LCSA results. If anything, it shows that already the consideration of one sustainability dimension for a PSB enabled the support of the decision-making process on the improvement actions, under the conditions of costs constraints. The augmentation of the operational LCSA framework with the PSB is therefore seen as a valid step towards enhancing the usability of LCSA at an automotive company.

4.3 Limitations and future research needs

As this study focused on overcoming the challenge of decision support within LCSA, the other methodical challenges like the maturity of S-LCA or the meaningful integration of LCC in the framework remain open points. To operationalize the LCSA framework, the company perspective was chosen for the life cycle analyses. The issue of trade-offs within as well as between LCSA dimensions was simplified by assuming equal weighting and a weak interpretation of sustainability. In addition, the focus of the LCSA analysis was laid on the production phase. These assumptions limit the ability of the presented approach to assess overall product sustainability improvements in a comprehensive way. Further research could address at integrating the LCC approach as suggested by Wood and Hertwich (2013) and at consolidating the product sustainability impacts as evaluated from a company perspective with those of the perspectives of other stakeholders. The LCSA implementation scheme and monetary valuation of customer relevant product sustainability features in this study was aimed at improving impacts in the production phase. An expansion to include and evaluate the impacts on the use phase in combination with the Product Sustainability Budget would be a next step to fully lift the presented approach to an analysis encompassing the entire life cycle. The social aspects were not considered in the WTP study in this research. This was primarily due to the fact that KO criteria from the company's point of view—like human rights violations—should not be monetized. Furthermore, the approach focused on customer relevant aspects of product sustainability in a car. The social impacts were deemed to be less customer relevant than the environmental aspects as the former cannot be made tangible that easily in a car. The WTP analysis was done for the German market; thus, studies in other markets would add to the understanding of WTP in different markets. As the WTP study quantified a hypothetical amount that consumers would be willing to pay, the development of the Product Sustainability Budget based on existing capitalized product sustainability features would ground it in tangible values. The application of the Product Sustainability Budget in a real case study should additionally test its viability. Finally, the question of how to weight sustainability dimensions would be a question that has to be answered for every company, individually. Furthermore, another approach like the valuation of costs incurring to a company due to worsened environmental impacts would add to the evaluation done in this paper. An investigation linked with avoided costs models (de Groot et al. 2002; de Groot 2006) could yield additional insights.

5 Conclusions

The goal of this paper was to develop and demonstrate a way to create a Product Sustainability Budget that would enable decision support in trade-off situations within an operationalized LCSA framework at an automotive company where impacts on the economic dimension would worsen, i.e., increase in costs. It could be shown that by applying a monetary valuation approach to measure the WTP of consumers for directly perceivable product features, a business case for a sustainability interior package for a car can be made. The study combined CBCA to find out a target group within a sample for the German premium car segment that would prefer the sustainability package to other interior packages and consider it in a purchase decision. In combination with an Advanced-Van-Westendorp analysis, the ideal price at which to sell this package could be determined. The share of the target group in the entire sample in combination with the derived selling price was fed into a business case calculation that could determine the expected profit from selling a sustainability interior package. The proposed Product Sustainability Budget would divert part of this profit towards measures for improving product sustainability. In this way, the most effective set of measures for improving overall product sustainability could be identified with an LCSA framework approach as long as the Product Sustainability Budget could compensate for an increase in costs. As this budget has been defined beforehand and is financed by product sustainability features, the argument for its adoption at a company should be facilitated. As the operationalized LCSA framework used weights to calculate an overall LCSA impact score, it allowed for trade-offs between sustainability dimensions and therefore represented the weak interpretation of sustainability. This could, in turn, harbor the risk for the company that uses this approach to be accused of green washing because improvement measures can be implemented that lead to a deterioration in one of the sustainability dimensions. Even though extreme social violations—like human rights violations—were categorized as KO criteria, other impacts—like worse remuneration of workers—could still be a result when applying the presented approach. Future research could look at the customer added value of improved social impacts, e.g., acquiring and labeling materials and components after the "Fair Trade" system. Furthermore, the customer relevant aspects of the use phase should be investigated. The additional consideration of the social sustainability dimension in future WTP studies might lead to a stronger PSB. More specifically, if future studies find additional WTP of customers for social sustainability in a car, the financial leeway to counter cost constraints would become even larger, enabling more product sustainability improvement measures to be realized at a company. Future research should also look at expanding the perspective within the operationalized approach to other stakeholders and the use phase. Furthermore, applying a cost approach for monetary valuation, e.g., avoided costs models, and deriving Product Sustainability Budgets from already existing business cases, or real figures and calculations of the sustainability package, would add to the findings of this paper.

Next steps towards improving the presented approach could be to (1) apply the strong interpretation of sustainability in the LCSA framework, i.e., not allowing for trade-offs between sustainability dimensions, (2) devise a sustainability interior package that also features the improvement of social impacts, (3) carry out a WTP study for this interior package in additional markets, (4) investigate how to include the entire vehicle life cycle in the assessment, and (5) develop a sustainability interior package that is actually offered to customers to determine the effective profit that is reaped through product sustainability improvement.

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References

Achabou MA, Dekhili S (2013) Luxury and sustainable development: is there a match? J Bus Res 66:1896–1903

Achtnicht M (2012) German car buyers' willingness to pay to reduce CO2 emissions

Backhaus K, Erichson B, Plinke W, Weiber R (2011a) Multivariate Analysemethoden. Eine anwendungsorientierte Einführung, 13th edn. Springer-Verlag, Berlin, Heidelberg

Backhaus K, Erichson B, Weiber R (2011b) Fortgeschrittene multivariate Analysemethoden. Eine anwendungsorientierte Einführung, 1st edn. Springer-Verlag, Berlin, Heidelberg

Backhaus K, Voeth M, Sichtmann C, Wilken R (2005) Conjoint-Analyse versus direkte Preisabfrage zur Erhebung von Zahlungsbereitschaften: eine modifizierte Replikationsstudie. Die Betriebswirtschaft DBW 65:439–457

Baier D, Brusch M (eds) (2009) Conjointanalyse. Springer Berlin Heidelberg, Berlin, Heidelberg

Baumgartner B, Steiner WJ (2009) Hierarchisch bayesianische Methoden bei der Conjointanalyse. In: Conjointanalyse. Springer Berlin Heidelberg, Berlin, Heidelberg, pp 147–159

Baxter T, Bebbington J, Cutteridge D, Harvey G (2003) The sustainability assessment model (SAM): measuring sustainable development performance. Offshore Europe. Society of Petroleum Engineers, In

Bebbington J, Brown J, Frame B (2007) Accounting technologies and sustainability assessment models. Ecol Econ 61:224–236

Behzadian M, Kazemzadeh RB, Albadvi A, Aghdasi M (2010) PROMETHEE: a comprehensive literature review on methodologies and applications. Eur J Oper Res 200:198–215

Ben-Akiva M, Boccara B (1995) Discrete choice models with latent choice sets. Int J Res Mark 12:9–24

Benayoun R, Roy B, Sussmann B (1966) ELECTRE: une méthode pour guider le choix en présence de points de vue multiples SEMA

Biesalski & Company (2014) Wertschöfungsreport Nachhaltigkeit 2014 - Was ist gutes Gewissen wert? In: 1–26

Biswas A (2016) A study of consumers' willingness to pay for green products. J Adv Manag Sci:211–215

Brans JP (1982) L'ingénièrie de la décision; Elaboration d'instruments d'aide à la décision. La méthode PROMETHEE. In: Nadeau R, Landry M (eds) L'aide à la décision. Nature, Instruments et Perspectives d'Avenir. Presses de l'Université Laval, Québec, pp 183–213

Cai Z, Aguilar FX (2013) Meta-analysis of consumer's willingness-to-pay premiums for certified wood products. J For Econ 19:15–31

Ciravegna L, Romano P, Pilkington A (2013) Outsourcing practices in automotive supply networks: an exploratory study of full service vehicle suppliers. Int J Prod Res 51:2478–2490

Cohen S (1997) Perfect union. CBCA marries the best of conjoint and discrete choice models. Mark res. Spring 9:12–17

Davies GR (2013) Appraising weak and strong sustainability: searching for a middle ground. Consilience. J. Sustain Dev 10(1):111–124

de Groot R (2006) Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. Landsc Urban Plan 75:175–186

de Groot RS, Wilson MA, Boumans RM (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41:393–408

Ding M, Eliashberg J, Huber J, Saini R (2005) Emotional bidders—an analytical and experimental examination of consumers' behavior in a Priceline-like reverse auction. Manag Sci 51:352–364

European Commission (2005) ExternE Externalities of Energy - Methodology 2005 Update

European Commission (2007) Setting emission performance standards for new passenger cars as part of the Community's integrated approach to reduce CO2 emissions from light-duty vehicles

European Commission (2008) Special Eurobarometer 295 - Attitudes of European citizens towards the environment

European Commission (2013) Flash Eurobarometer 367 - Attitudes of Europeans Towards Building the Single Market for Green Products

Figge F (2005) Capital substitutability and weak sustainability revisited: the conditions for capital substitution in the presence of risk. Environ Values 14:185–201

Figge F, Hahn T (2004) Sustainable value added - measuring corporate contributions to sustainability beyond eco-efficiency. Ecol Econ 48: 173–187

Figge F, Hahn T (2005) The cost of sustainability capital and the creation of sustainable value by companies. J Ind Ecol 9:47–58

Finkbeiner M, Reimann K, Ackermann R (2008) Life cycle sustainability assessment (LCSA) for products and processes. In: SETAC Europe 18th Annual Meeting, 25–29 May. Warsaw, Poland

Finkbeiner M, Schau EM, Lehmann A, Traverso M (2010) Towards life cycle sustainability assessment. Sustainability 2:3309–3322

Fontes J, Bolhuis A, Bogaers K et al (2015) Handbook for Product Social Impact Assessment, 2.0. PRé Sustainability, Amsterdam

Fontes J, Tarne P, Traverso M, Bernstein P (2016) Product social impact assessment. Int J life cycle assess

Frame B, Cavanagh J (2009) Experiences of sustainability assessment: an awkward adolescence. Account Forum 33:195–208

Gelman A, Carlin J, Stern H, Rubin D (2004) Bayesian Data Analysis, 2. CRC Press, Boca Raton, Florida

Govindan K, Jepsen MB (2016) ELECTRE: a comprehensive literature review on methodologies and applications. Eur J Oper Res 250:1–29

Groening C, Inman JJ, Ross WT (2015) The role of carbon emissions in consumer purchase decisions. Int J Environ Policy Decis Mak 1:261

Ha-Brookshire JE, Norum PS (2011) Willingness to pay for socially responsible products: case of cotton apparel. J Consum Mark 28: 344–353

Hahn T, Figge F, Barkemeyer R, et al (2013) Sustainable value in automobile manufacturing - an analysis of the sustainability performance of automobile manufacturers worldwide. Sustainable Value Research, Marseille and Leeds

Hetterich J, Bonnemeier S, Pritzke M, Georgiadis A (2012) Ecological sustainability – a customer requirement? Evidence from the automotive industry. J Environ Plan Manag 55:1111–1133

Johnson R, Orme B (1996) How Many Questions Should You Ask in Choice-Based Conjoint Studies? Sawtooth Softw Research P:23

Jørgensen A, Hermann I, Mortensen J (2010) Is LCC relevant in a sustainability assessment? Int J Life Cycle Assess 15:531–532

JRC (2012) Towards a Life-Cycle Based European Sustainability Footprint Framework. Publications Office of the European Union, Luxembourg

Kagel J (1995) Auctions: A Survey of Experimental Research. In: Kagel J, Roth A (eds) The Handbook of Experimental Economics. Princetown University Press, Princetown, pp 501–585

Kaltenborn T, Fiedler H, Lanwehr R, Melles T (2013) Conjoint-analyse. In: Spieß M, Matiaske W (eds) Sozialwissenschaftliche Forschungsmethoden, Band 7, 1st edn. Rainer Hampp Verlag, Munich and Mering, p 143

Karlewski H (2016) Social life cycle assessment in the automotive industry. Technische Universität Berlin, Berlin

Klöpffer W (2003) Life-cycle based methods for sustainable product development. Int J Life Cycle Assess 8:157–159

Klöpffer W (2008) Life cycle sustainability assessment of products. Int J Life Cycle Assess 13:89–95

Klöpffer W, Grahl B (2014) From LCA to sustainability assessment. In: Life cycle assessment (LCA). Wiley-VCH Verlag GmbH & Co. KGaA, Weinheim, pp 357–374

Krupa JS, Rizzo DM, Eppstein MJ, Brad Lanute D, Gaalema DE, Lakkaraju K, Warrender CE (2014) Analysis of a consumer survey on plug-in hybrid electric vehicles. Transp Res Part A Policy Pract 64:14–31

Kruschke JK, Vanpaemel W (2015) Bayesian estimation in hierarchical models. In: Busemeyer JR, Wang Z, Townsend JT, Eidels A (eds) The Oxford handbook of computational and mathematical psychology. Oxford University Press, Oxford, pp 279–299

Lankoski L (2010) Customer willingness to pay for sustainability in the food sector: An examination of three WTP types

List JA, Gallet CA (2001) What experimental protocol influence disparities between actual and hypothetical stated values? Environ Resour Econ 20:241–254

Liu Y (2014) Household demand and willingness to pay for hybrid vehicles. Energy Econ 44:191–197

Markandya A (2012) Externalities from electricity generation and renewable energy, methodology and application in Europe and Spain. Cuad Económicos ICE 83:85–100

Martínez-Blanco J, Lehmann A, Muñoz P, Antón A, Traverso M, Rieradevall J, Finkbeiner M (2014) Application challenges for the social life cycle assessment of fertilizers within life cycle sustainability assessment. J Clean Prod 69:34–48

Meadows D, Meadows D, Randers J, Behrens W (1972) The limits to growth. Universe Books

Meadows D, Randers J, Meadows D (2007a) Grenzen des Wachstums das 30-Jahre update - signal zum Kurswechsel, 2. Auflage. S. Hirzel Verlag, Stuttgart

Meadows D, Randers J, Meadows D (2007b) Limits to Growth - the 30 Year Update Earthscan, London, United Kingdom

Müller H (2008) Empirische Untersuchung zur Messung der Preiswahrnehmung mittels Pricesensitivity-meter. Otto-von-Guericke-Universität, Magdeburg

Murphy JJ, Allen PG, Stevens TH, Weatherhead D (2005) A meta-analysis of hypothetical Bias in stated preference valuation. Environ Resour Econ 30:313–325

Natural Capital Coalition (2016) Natural Capital Protocol

Neugebauer S, Forin S, Finkbeiner M (2016) From life cycle costing to economic life cycle assessment—introducing an economic impact pathway. Sustainability 8:428

Opricovic S, Tzeng G-H (2004) Compromise solution by MCDM methods: a comparative analysis of VIKOR and TOPSIS. Eur J Oper Res 156:445–455

Opricovic S, Tzeng G-H (2007) Extended VIKOR method in comparison with outranking methods. Eur J Oper Res 178:514–529

Ottman JA, Stafford ER, Hartman CL (2006) Avoiding Green Marketing Myopia. Environment 48:22

Parguel B, Benoît-Moreau F, Larceneux F (2011) How sustainability ratings might deter “greenwashing”: a closer look at ethical corporate communication. J Bus Ethics 102:15–28

PUMA (2011) PUMA’s Environmental Profit and Loss Account for the year ended 31 December 2010

Reinecke S, Mühlmeier S, Fischer P (2009) Die van Westendorp-Methode: Ein zu Unrecht vernachlässigtes Verfahren zur Ermittlung der Zahlungsbereitschaft? Wirtschaftswissenschaftliches Stud 38(2):97–100

Roll O, Pastuch K, Buchwald G (2012) Praxishandbuch Preismanagement: Strategien-management-Lösungen. John Wiley & Sons

Sattler H, Nitschke T (2003) Ein empirischer Vergleich von Instrumenten zur Erhebung von Zahlungsbereitschaften. Schmalenbachs Zeitschrift für betriebswirtschaftliche Forsch ZfbF 55:364–381

Sichtmann C, Stingel S (2007) Limit conjoint analysis and Vickrey auction as methods to elicit consumers’ willingness-to-pay. Eur J Mark 41:1359–1374

Simon H, Fassnacht M (2009) Preismanagement. Gabler Verlag, Wiesbaden

Singh RK, Murty HR, Gupta SK, Dikshit AK (2009) An overview of sustainability assessment methodologies. Ecol Indic 15:281–299

Skiera B, Gensler S (2002) Berechnung von Nutzenfunktionen und Marktsimulationen mit Hilfe der Conjoint-Analyse, Teil I. Wirtschaftswissenschaftliches Stud 31(4):200–206

Stanton EA (2012) The tragedy of maldistribution: climate, sustainability, and equity. Sustainability 4:394–411

Sundt S, Rehdanz K (2015) Consumers’ willingness to pay for green electricity: a meta-analysis of the literature. Energy Econ 51:1–8

Tarne P, Lehmann A, Finkbeiner M (2018) Introducing Weights to Life Cycle Sustainability Assessment – How Do Decision Makers Weight Sustainability Dimensions?

Tarne P, Traverso M, Finkbeiner M (2017) Review of life cycle sustainability assessment and potential for its adoption at an automotive company. Sustainability 9:670

Traverso M, Finkbeiner M, Jørgensen A, Schneider L (2012) Life cycle sustainability dashboard. J Ind Ecol 16:680–688

Tully SM, Winer RS (2013) Are people willing to pay more for socially responsible products: a meta-analysis. SSRN Electron J. https://doi.org/10.2139/ssrn.2240535

van Westendorp P (1976) NSS Pricesensitivity-meter (PSM) – a new approach to study consumer perception of prices. In: Proceedings of the 29th ESOMAR Congress, Venice, 5–9 September 1976, pp 139–167

Veisten K (2007) Willingness to pay for eco-labelled wood furniture: choice-based conjoint analysis versus open-ended contingent valuation. J For Econ 13:29–48

Voeth M, Niederauer C (2008) Ermittlung von Preisbereitschaften und Preisabsatzfunktionen. In: Handbuch Marktforschung: Methoden, Anwendungen, Praxisbeispiele. Gabler Verlag, Wiesbaden, pp 1073–1095

Völker R, Tachkov P (2013) Der Wert nachhaltiger Produkteigenschaften. Bestimmung der Zahlungsbereitschaft für umweltfreundliche Güter Ökologisches Wirtschaften 28:46–59

Wood R, Hertwich EG (2013) Economic modelling and indicators in life cycle sustainability assessment. Int J Life Cycle Assess 18:1710–1721

2.4 Combination of Results to Increase the Applicability of LCSA

The individual research objectives of this thesis (section 1.5) have been addressed in individual publications in sections 2.1–2.3. This section presents how the combination of these results helped in increasing the applicability of LCSA by filling the remaining gaps of the operationalized framework as shown in section 2. The results and findings from section 2.1.2 showed that MRIO analysis was not deemed suitable to streamline primary data collection in S-LCA through social hotspot assessment. Therefore, a material-based approach to assess the social risk related to material production was devised (c.f. section 2.1.3), which can feed into the LCSA framework at the component level to give an indication on the social impacts associated with a components manufacture. Therefore, the gap regarding the quantitative assessment of the social dimension in Table 1 could be filled. Table 7 shows the operationalized LCSA framework with the enabled S-LCA dimension.

Table 7: Operationalized LCSA framework after the selection of social indicators and the development of an approach for social hotspot analysis that has been applied to the component level of a vehicle.

Component	**LCA Criticality Points**	**LCC Criticality Points**	**S-LCA Criticality Points**	**Overall LCSA Criticality Points**
A	72	11	**100**	?
B	82	17	**42**	?
C	20	52	**45**	?
...	...	...	...	...

The study on weighting in LCSA suggested to adopt the weightings made in the decision process of the company (c.f. section 2.2). The weights for the LCSA dimensions were identified via Limit Conjoint Analysis and were applied to the operationalized LCSA framework to enable the aggregation of the criticality points. Following the results of section 2.2, the LCA criticality points were considered with 35%, the LCC criticality points with 34% and the S-LCA criticality points with 31% for the overall LCSA criticality points. The introduction of these weights enabled a combined assessment within the operationalized LCSA framework, i.e. the calculation of overall LCSA criticality points. Table 8 shows the operationalized framework after the quantification of the S-LCA dimension and the introduction of weights.

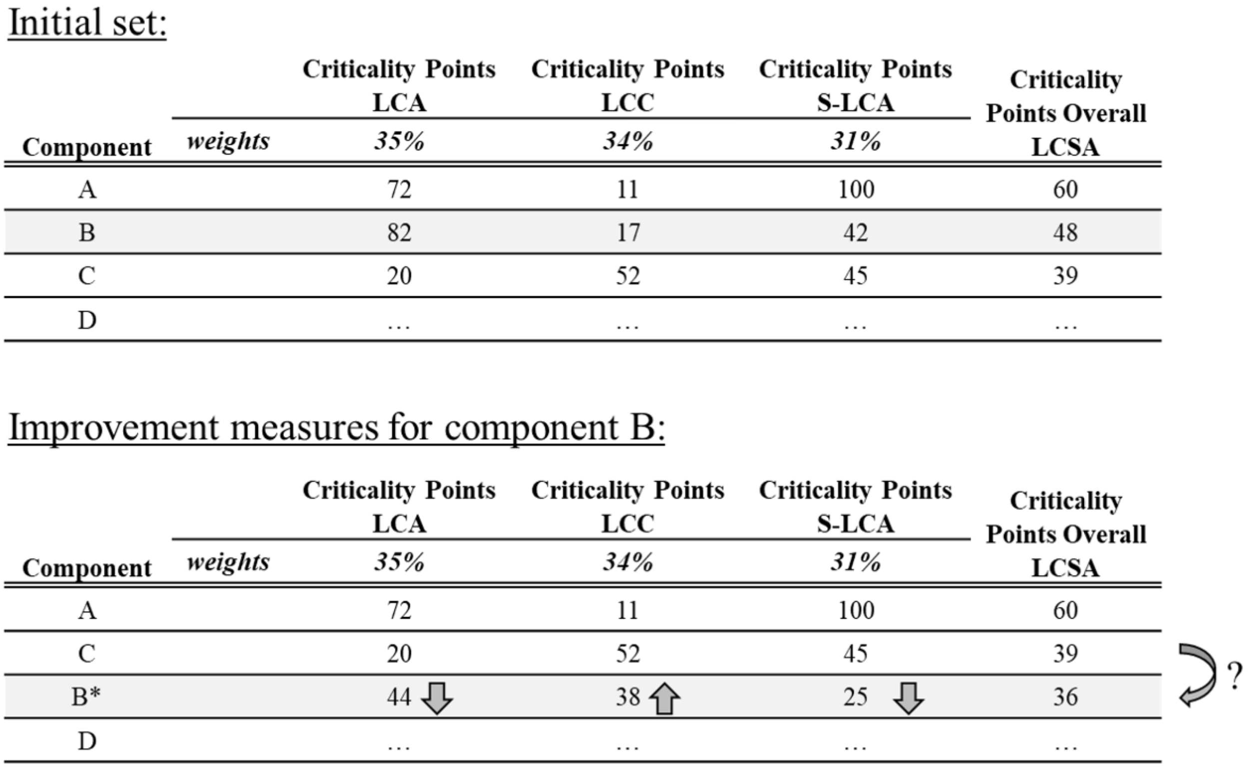
Table 8 shows that the weighting set, which was derived by the third research paper, enabled a clear interpretation of results for decision makers as it made the calculation of overall LCSA criticality points possible. When looking at this combined score, the most critical components in terms of overall product sustainability impacts can be identified — in Table 8 the most critical component is component A. It can be seen that this high criticality score arises from relatively high environmental and social impacts, whereas the economic impacts are comparatively low. This way, decision makers

Table 8: Operationalized LCSA framework after the introduction of weights

		LCA Criticality Points	**LCC Criticality Points**	**S-LCA Criticality Points**	**Overall LCSA Criticality Points**
Component	***weights***	***35%***	***34%***	***31%***	
A		72	11	100	**60**
B		82	17	42	**48**
C		20	52	45	**39**
...		...	...	...	**...**

can get a quick overview of the most critical components and from which sustainability dimension this criticality stems from.

After the introduction of weights, the LCSA framework was basically ready for use at an automotive manufacturer. But in practice an improvement measure for a component for an overall improvement in terms of product sustainability may not be implemented based on the existing decision processes at the company. As the decision process is still more strictly geared towards business cases, the economic dimension is in practice treated with higher priority as the results of the weighting study would suggest. If the improvement measure can only be implemented with a simultaneous increase in costs, it will therefore most likely not be realized. Figure 7 shows the challenge that remains when applying the operationalized LCSA framework within a company.



Initial set:

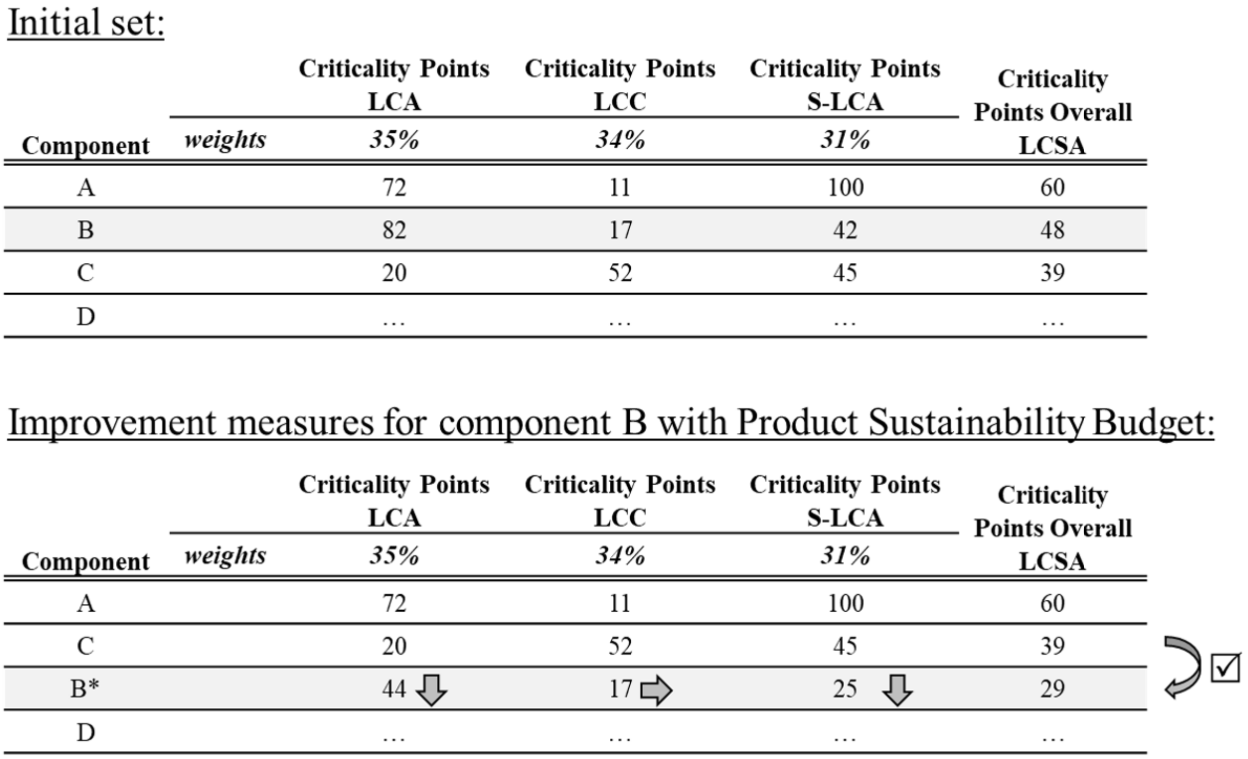
Component	weights	Criticality Points LCA	Criticality Points LCC	Criticality Points S-LCA	Criticality Points Overall LCSA
		35%	*34%*	*31%*	
A		72	11	100	60
B		82	17	42	48
C		20	52	45	39
D		…	…	…	…

Improvement measures for component B:

Component	weights	Criticality Points LCA	Criticality Points LCC	Criticality Points S-LCA	Criticality Points Overall LCSA
		35%	*34%*	*31%*	
A		72	11	100	60
C		20	52	45	39
B*		44 ⇩	38 ⇧	25 ⇩	36
D		…	…	…	…

Figure 7: Remaining challenge for the implementation of LCSA into decision making

As shown in Figure 7, the impacts of component B would change due to an improvement measure (B*). The environmental (LCA) and social (S-LCA) dimension have improved from the company's perspective while the impact on the economic (LCC) dimension has deteriorated. By applying the newly elicited weighting set, the overall change in criticality would be acceptable (the overall LCSA criticality points of B* are lower than those of B). The increase in costs, however, would most probably constitute a deal-breaker in a business context. Thus, the decision for improving component B is still uncertain in a company context. The solution proposed in this thesis to address this challenge is the PSB (c.f. section 2.3). Figure 8 shows how the challenge of implementing LCSA into decision making is solved by the introduction of the PSB.



Initial set:

Component		Criticality Points LCA	Criticality Points LCC	Criticality Points S-LCA	Criticality Points Overall LCSA
	weights	*35%*	*34%*	*31%*	
A		72	11	100	60
B		82	17	42	48
C		20	52	45	39
D		…	…	…	…

Improvement measures for component B with Product Sustainability Budget:

Component		Criticality Points LCA	Criticality Points LCC	Criticality Points S-LCA	Criticality Points Overall LCSA
	weights	*35%*	*34%*	*31%*	
A		72	11	100	60
C		20	52	45	39
B*		44 ⇩	17 ⇨	25 ⇩	29
D		…	…	…	…

☑

Figure 8: Linking of the Product Sustainability Budget to the LCSA framework enables the clear decision support in trade-off situations where costs would rise

As can be seen in Figure 8, when the PSB is applied, the impact on the economic dimension can be compensated for. The LCC criticality points of component B* went up from 17 to 38 in the LCSA framework without PSB (c.f. Figure 7) meaning that the costs for component B* had increased. When the cost increase for component B* lies within the economic boundary of the PSB, the impact on the LCC dimension is neutralized (the LCC criticality points of component B* remain at 17). Therefore, the assessment of the improvement measure for component B can be clearly supported in a company context.

If several improvement measures are available, one disadvantage is that with a different LCA or S-LCA performance and different costs, the optimal overall solution is not visible. Therefore, a complementary approach is suggested. It is proposed by this research that the PSB is applied to define the economic dimension of a solution space within which the social and environmental impacts are optimized. This way, out of a set of different possible measures to improve products sustainability, the set of measures which satisfies all dimensions of the solution space can be identified. That means, the set of measures that minimizes social and environmental impact the most while staying within the defined economic boundaries. Therefore, the PSB can be used to identify the optimal set of improvement measures. An example to demonstrate this approach is presented in the following. Table 9 lists several theoretical measures to improve product sustainability impacts.

Table 9: Theoretical measures to improve product sustainability impacts. The table contains fictional values.

Improvement measure	Costs	Criticality Points LCA	Criticality Points S-LCA	Total	Criticality Points / €
I	6.70 €	-30	-17	-47	-7.0
II	3.40 €	-10	-12	-22	-6.5
III	4.20 €	-15	-15	-30	-7.1
IV	5.00 €	-25	-10	-35	-7.0
V	2.10 €	-9	-4	-13	-6.2
VI	6.00 €	-25	-19	-44	-7.3

To find the ideal solution, linear optimization like Excel Solver can be used. With a PSB of 10 €as a constraint, the sum of LCA and S-LCA criticality points is optimized. This gives as a result that the combination of improvement measure II and VI would yield the highest improvement in product sustainability (-66 criticality points) while still staying within the limit of the PSB (9.40 €).

3 Discussion

In this section, the key findings of this thesis (section 3.1) and their transferability (section 3.2) are discussed. Furthermore, the limitations and challenges are discussed (section 3.3).

3.1 Key Findings

In section 2.4 it was shown how the combination of the individual findings, as presented in section 2.1–2.3, contributed to increasing the applicability of LCSA. Through the combined findings progress was made to support practitioners at an automotive company to:

a) identify the components of a vehicle which have the highest LC sustainability impacts and therefore the greatest improvement potential,

b) evaluate the overall impact of improvement measures for components on LC sustainability impacts in case of trade-offs by using a weighting set and

c) facilitate the implementation of LCSA results into the product-related decision process by buffering an increase in cost via the PSB That means that the approach enables practitioners to also implement the previously identified product sustainability improvement potential at an automotive company. Therefore, the developed approach is practically able to fulfill the initially set goal of this thesis (c.f. section 1.5).

The presented approach in this thesis pursued the practical implementation of LCSA at an automotive company by addressing key challenges (Tarne et al., 2017). Especially within the automotive sector, there has not been a similar approach. One known approach to address all three dimension of a product?s impact was the PSI by Ford (Schmidt & Taylor, 2006). However, the PSI was developed at a very rudimentary level, addressing five environmental, two social and one economic indicator. The index has also not substantially been developed further since its inception. The SEEBalance approach by BASF (Saling et al., 2002, 2005) can be seen as a practical implementation of LCSA in the chemical industry. The latest update added a social dimension to the already existing environmental and economic assessment (Kölsch et al., 2008). It could therefore be counted to the approaches that extended an eco-efficiency analysis by a social dimension (Klöpffer & Grahl, 2014). In contrast to SEEBalance, the method proposed in this thesis is tackling the life cycle based assessment of a more complex product as well as the implementation of those results into product-related decision making. To the author’s knowledge, there exists no comparable approach to implementing LCSA results in company decision processes like the PSB that has been developed in this thesis. Most research is looking at the valuation of externalities (Jasinski et al., 2015) rather than the possible benefits of introducing product sustainability. This is not implying that the valuation of externalities is an invalid route to monetization. However, as the externalities of the automotive industry do not seem to be internalized in the near future, they are not to be considered (c.f. section 1.3.2). Consequently, these costs will not be accepted by decision makers at a company because they will effectively not arise.

From a practitioner's point of view, the combination of the findings on S-LCA (section 2.1) and the integration of LCSA results into decision making processes at an automotive company (section 2.3) are crucial to receive an operational LCSA framework. The introduction of weights (section 2.2) can be seen as a facilitator for LCSA-based decision making, which is not integral to the integration of LCSA based results into decision making but rather improves the interpretation of results. The clear interpretation of LCSA results to support decision makers who are unfamiliar with LCSA results is a much needed gap to fill to increase the applicability of LCSA from a practitioner's point of view (Tarne et al., 2017). To introduce weights to address this gap is not an innovation as many researchers have done so, e.g. Ghadimi et al. (2012), Manzardo et al. (2012), Wang & Chan (2013) and Ren et al. (2015). The introduction of weights might be seen as critical as it might lead to LCSA practitioners presenting decision makers only with the aggregated results, while it is an important feature to still provide the underlying disaggregated results (Traverso et al., 2012). So it appears that the step of introducing weights to LCSA is not necessary to ensure the minimum of applicability needed for adoption at an automotive company. However, the introduction of weights added an additional information that greatly helps in supporting decision making while the original data can still be presented — as shown in Table 8. It is therefore still seen as an integral part towards the overarching goal — increasing the applicability of LCSA.

3.2 Transferability of Results

In principle, the logic of the relative evaluation approach at the component level is not limited to the application within the automotive industry but can be transferred to any other context. The application of the assessment logic to the component level would depend on the type of product. For a material, for instance, this granularity would not be expedient. While for more complex products, e.g. like buildings, airplanes, smartphones etc., it would be feasible. The indicators that have been chosen for the operationalized LCSA framework in this thesis would be dependent on the respective context. In principle, the way of increasing the applicability of LCSA as developed in this dissertation could be applied at any company in any industry sector. The approach to assess the social impacts is not limited to the use within the automotive industry. The material-based social hotspot assessment followed a rationale that is independent from a product. Its logic and the subsequent integration into the LCSA framework can be applied anywhere. However, depending on the industry that wants to apply this approach, the scope would have to be expanded to cover the relevant materials used by the respective industry. The social indicators will be dependent on the company and the context, respectively, where the approach is applied. The logic how to identify relevant social topics and indicators can be taken from this thesis and applied at any company. The relative evaluation is not a new concept within sustainability assessment in general and LCSA in particular (Ghadimi et al., 2012; Manzardo et al., 2012; Wang & Chan, 2013; Ren et al., 2015; Tarne et al., 2017). The logic of relative evaluation can be applied in every industry or to every kind of product.

The application of Limit Conjoint Analysis in this thesis was the first time that this method was applied to derive weights for LCSA. In contrast to more prominently used approaches — like the Analytical Hierarchy Process (Saaty, 1990) — this method more closely resembles the decision process in a purchase decision. This method can be applied in any industry to elicit weights for the LCSA dimensions as desired. The weighting of sustainability dimensions at another company could be determined in the same way as was done in this thesis. However, the results of the Limit Conjoint Analysis of this thesis cannot be transferred to other products or even other companies within the same industry. Because they reflect the trade-off decision process within the investigated company, which will most certainly differ in every company. As already discussed in Tarne et al. (2018b), the introduction of weights and aggregation to an overall score constitutes the adoption of the weak interpretation of sustainability. This was a deliberate decision as this approach was deemed to garner more support within the industry as compensations or trade-offs between dimensions are possible (Davies, 2013). In principle, the concept of strong sustainability can also be translated into this evaluation (Tarne et al., 2018b).

The principle of the PSB as introduced in this dissertation could be applied to any product. However, the results of the specific study (c.f. section 2.3) cannot be transferred as they have been derived from a specific study setup and sample. The type of study and the kind of product sustainability features that bring a customer added value will also depend strongly on the industry in question and how sustainability of the product can be made apparent to the consumer or make it even an experience.

3.3 Limitations & Challenges

There still remained several limitations and challenges that impacted the degree of applicability of the LCSA framework. The methodological limitations and challenges are discussed in section 3.3.1, the practical limitations and challenges in section 3.3.2.

3.3.1 Methodological Limitations and Challenges

The direction of this thesis was determined by the results of structured literature review of the state of the art of LCSA in the automotive industry. Possible shortcomings in that analysis might have led to blind spots in the presented LCSA approach. Therefore, the operationalized LCSA framework in this thesis might be missing relevant aspects.

The proposed approach to increase the applicability of the LCSA framework expresses the relative performance of an individual component over a vehicle's life cycle compared to the other components of a vehicle. That means that only relative assessments within the assessed product, about e.g. whether a change in a component improves or worsens its overall impact, can be made. This focus on relative evaluation is in line with the definition of product sustainability assessment as introduced in section 1.1. Assessments in relation to absolute values, e.g. how far a component or the entire vehicle is away from a certain goal, cannot readily be drawn from this approach. The information, however, is contained in the early steps of the method and could be fed into a differently set up LCSA framework.

The approach presented in this thesis applies allocation of LCSA impacts of the entire vehicle's LC via the weight of the components. This constitutes a possible source for inaccuracies and suboptimal decision making. The Global Warming Potential (GWP) of a component's production, for example, can be determined for every component as their material composition and production processes are known. Regarding the use phase, not every component will have the same impact, however. An allocation of the use phase emissions by component weight was used in this thesis which might fall short as other properties like aerodynamics or rolling resistance also have an influence on the fuel economy of a vehicle (Reynolds & Kandlikar, 2007). Apart from determining the effect a component has on the consumption of a vehicle, the effect on other impacts in the use phase would be far harder to determine, e.g. land use or the social impacts. The possible distortion generated by the mass allocation applied in the LCSA framework might be exacerbated in the future with the increase in electrified vehicles. With the use of regenerative braking systems, the mass of a vehicle becomes even less of an impact factor on its energy consumption in the use phase because kinetic energy is recuperated while braking (Redelbach et al., 2012; Brooker et al., 2013).

The selection of social topics and indicators was based on the materiality matrix of an automotive company. To identify the social topics of relevance, a threshold value of seven was chosen. A lower or higher threshold value might have resulted in more or fewer social topics and indicators, respectively. It might be debatable, whether S-CLA, and thus LCSA, has been made operational by the applied approach in this thesis. With the material-based risk assessment an intermediate solution of an impact assessment for material supply chains was introduced. But technically, the applied approach is a **risk assessment** approach that has been implemented in the applied LCSA framework. That means that social risks have been treated like social impacts and that the challenge of data acquisition for impact assessment in S-LCA remains. Karlewski (2016) found one of the main challenge of S-LCA to retrieve the company-specific social data along the life cycle. This has therefore been confirmed again by this thesis. This research found that MRIO databases do not offer substantial support for conducting social hotspot analysis. This seems to be contrary to the trend of increasingly using MRIO analysis to build up social risk databases (Benoit-Norris et al., 2012; Benoît Norris, 2014; Ciroth & Eisfeldt, 2015). When these approaches are looked at more closely, the degree of the application of MRIO is more differentiated. For the construction of the SHDB, for instance, the Eora database was used to establish a worker hour model that made it possible to estimate the number of hours of labour that happen in a respective region and industry sector (Benoit-Norris et al., 2012). Then it was determined how many of these hours had a potential risk of being affected by social violations. This was done by resorting to data by third parties — like the Worldbank or the International Labour Organization — none of which are MRIO databases. Therefore, the use of MRIO was only limited to the identification of the amount of "risk hours" within a certain region and industry sector. The risk of a social violation within that region and industry sector were determined by relying on other sources. A material-based social risk assessment was proposed that enabled S-LCA to a sufficient degree within the company. This approach only evaluated the social risks that are associated with the material production at a preliminary level. Any additional production steps necessary to produce a component are neglected in this risk assessment. This might lead to crucial risks being overlooked, especially as certain components, like electronics, might be not as risky from a material standpoint but manufacturing/assembly might be done under precarious circumstances (Chan & Pun, 2010). Also, this approach assumed that the general social conditions in a country define the social risk of an individual supply chain. It is also very possible that a company producing in a "high risk" country upholds the same "low risk" working conditions throughout all production sites and is therefore not contributing to any social risk within the supply chain. The opposite case is also possible, i.e. a company that is operating in a "low risk" country might violate the working conditions, therefore contributing to social risk within the supply chain. Furthermore, the material-related approach requires data on the worldwide production of said material. This is readily available for metals from the USGS (United States Geological Survey, 2013) and thus covers the majority of materials in a vehicle (BMW Group, 2014). However, the production information on plastics, the second largest material group in a vehicle, is not as readily available. In the material-based risk assessment approach it could be questioned why the definition of a very high risk of human rights violations was not consistently treated as a KO criterion (c.f. page 66). At the lower aggregation levels, i.e. single production steps, it was treated as a KO criterion but when the risk scores were aggregated to the final material risk score, this logic was dropped. This change in logic was introduced after the first evaluations, where the definition as KO criterion was also kept at the highest aggregation level, resulted in an overall material risk score of -2 for most materials. As the target of the method was to identify hotspots to prioritize primary data collection and enable quantitative evaluation of the social dimension within LCSA, these results did not bring about any benefit as no differentiation between materials was possible. That changed after removing the KO criterion in the last aggregation step. Which means, however, that the risk of human rights violations in the production countries of a material can be compensated for by less critical risks for other social topics. This mitigation of the evaluation logic was nevertheless accepted as it gives practitioners the possibility to focus on a few materials with high risk and actually be able to initiate next steps to improve transparency and remediating these risks. When no differentiation between materials is possible, the challenge of addressing all at once might prove to be a too sumptuous task or a focusing on a few material out of these might be perceived as arbitrary and thus lose credibility as a measure within a company. The presented approaches to identify social hotspots still only constitute the first step of an S-LCA. The social risk or social hotspot analysis functions as a guide as to where to focus the data collection efforts. That means, primary data has still to be gathered for the critical supply chains, in order to be able to carry out a substantiated S-LCA. The wish of parts of the S-LCA community to also include the assessment of positive impacts (Sala et al., 2015; Petti et al., 2018) has not been addressed by this research. The missing of a complete method definition of S-LCA, especially regarding the impact assessment (Arcese et al., 2018; Petti et al., 2018) was not seen as much as a challenge as the missing of a clear indicator set (Sala et al., 2015). In addition to the challenge of data acquisition and evaluating social impacts for S-LCA, it is still not clearly defined how social impacts in the use phase should be addressed and related to the functional unit, let alone for single components of a product. The integration of different affected stakeholders in different life cycle stages with respective social impacts remains a challenge.

The introduction of weights increased the applicability of the LCSA framework because the interpretation of the overall criticality of components and the impact of improvement measures can clearly be determined. It can be questioned, whether this approach of aggregation always leads to the favorable results of a society's perspective as trade-offs between LCSA dimensions are accepted. The weighting of other stakeholders will surely differ from the ones derived in this research. Another way of trying to achieve clear interpretation of LCSA results could have been monetization instead of weighting. As soon as all results have been translated into monetary terms, i.e. monetized, they can be aggregated as they are expressed in the same metric and trade-offs can be easily compared. This research decided to separate the interpretation of results on an aggregated basis and the relation of the sustainability impacts to monetary values for several reasons. One very simple reason was that it was deemed imprudent to put a price tag on social impacts, enabling the clear compensation of a human life with increased revenue. Another reason was that the concept of monetizing LCA impact category results is always combined with several uncertainties (e.g. reliability in elicited price levels for non-market goods like years of life). Furthermore, endpoint categories are often used for monetization of sustainability impacts, which are themselves determined based on several assumptions (Bare et al., 2000; Finnveden et al., 2009; Klöpffer & Grahl, 2014; Pizzol et al., 2015). The motivation of the monetization step in this research was to investigate the positive effects, i.e. the possible additional revenue, generated by product sustainability. It was expected that this would gain more attention and support within a company. From a methodological standpoint, it was also deemed more preferable to separate each step of the LCSA as proposed in this thesis:

1. Separate evaluation of impacts in every LCSA dimension
2. Aggregation and interpretation of overall LCSA result
3. Integration of results in the company decision process via application of the PSB

This way, it is possible to present decision makers with any desired level of detail of the results, increasing transparency of results and enabling more proficient decision makers to resort to additional information.

3.3.2 Practical Limitations and Challenges

The integration of the presented approach was carried out at the development department of an automotive company. One of the challenges for the LCSA practitioner regarding the execution of the presented approach is to coordinate the expertise for the assessment of the impacts of improvement measures on the different dimensions. Even if he or she manages the assessment independently, the results will most likely have to be synchronized with several departments as the responsibility for different topics is usually spread between different departments within an organization. As these separate competencies are usually strongly delineated within large organizations, following a so-called silo mentality (Stone, 2004), the results and proposed actions by the LCSA framework have to be acknowledged by the respective departments before they can be implemented in the decision processes within the company. Alignment between departments for setting up the LCSA framework would ease the evaluation, e.g. if social topics have been chosen according to the materiality matrix and confirmed by the strategy and purchasing department. However, the respective departments might still have their own way of analysis and may not welcome an external department assuming competencies that they see in their own area of responsibility.

One point that is stressed in the LCSA community is the requirement that the system boundaries of all three LC analyses have to be the same (Klöpffer, 2008; Klöpffer & Grahl, 2014). Or that one common life cycle inventory is compiled which is then used for all three (LCA, LCC, S-LCA) impact assessments (Zamagni, 2012). While this seems plausible from a methodological point of view, the experience of this research in a practical context was different. As long as the goal, scope, functional unit and cut-off criteria are the same for all three analyses, the results can be used for informing decision making, even though the system boundaries might vary from one assessment to the other. The practical experiences of this thesis therefore follow the line of Heijungs et al. (2009) who regard this as the natural result of three different perspectives on the same object of interest. The integration of LCSA-based results into the decision making process via the PSB presupposes that the potential profit from product sustainability features can actually be realized. Even though this research put forth strong evidence for such a business case, the reality might look different. Regarding the PSB, the take rate of a real sustainability interior package could deviate from the study results as the latter depend on several factors. For one, if the real sustainability interior package consisted of different components than the one defined in this thesis, the take rate could differ. The impact of overstated WTP, i.e. the effect that participants state a higher WTP because they think it is expected of them, might lie above or below the assumed ratios of Murphy et al. (2005) and List & Gallet (2001).

4 Conclusions & Outlook

In this section, the added scientific value of this dissertation is presented in section 4.1, while section 4.2 gives recommendations for future research. An overview of the essential themes of this thesis is given in Table 10.

Table 10: Overview of the main themes

	Key findings & added value	**Transferability of results**	**Methodological limitation & challenges**	**Practical limitation & challenges**	**Methodological steps forward**	**Practical steps forward**
Relative evaluation at component level	The approach enabled the identification of the component with the greatest product sustainability improvement potential within a vehicle. This kind of evaluation method has — to the author's knowledge — not been presented before.	The evaluation logic is not limited to the automotive industry but can be transferred to any other product that is sufficiently complex, i.e. consists of components.	The approach allocated the LCSA impacts of the entire vehicle's LC to the components by weight. This constitutes a possible source for inaccuracies and suboptimal decision making because not every component's impact on the use phase is a result of its weight.	Even though the greatest improvement potential at component level can be identified, a larger, because cross-cutting, improvement potential could be overlooked. Assessments in relation to absolute values, e.g. how far a component or the entire vehicle is away from a certain goal, cannot readily be drawn from this approach.	Further research should investigate the relationship between components and their sustainability impacts in the use phase of a vehicle.	Another approach to identify the greatest improvement potential might be to look at cross-cutting issues like the materials used in the entire vehicle. Also, an approach to steer product sustainability improvement measures towards absolute target values for vehicles, e.g. the 2°C target of the Paris Agreement, would be of interest to companies.
Social indicators	The applied method enabled the selection of social indicators relevant to the company and formed the basis for the assessment of the S-LCA dimension.	The logic how to identify relevant social topics and indicators can be applied at any company. The resulting indicators will most likely differ between companies.	To select social topics, the importance threshold of seven was chosen in the material matrix. A different threshold might yield additional or fewer indicators.	Regarding the identification of social topics and indicators relevant to the company, this thesis relied on sources from 2014.	Investigate the robustness of the selected indicator set by carrying out the same method while considering different importance threshold levels.	Next steps could look at, whether the identified social topics and indicators would still be relevant when considering more recent sources.
MRIO analysis	This research found that MRIO analysis did not offer substantial support for conducting social hotspot assessment.	The findings can be utilized by any practitioner who considers MRIO databases for SC analysis.	The analysis of MRIO databases was based on a unified form. Therefore, the results might be distorted as some databases had to be aggregated. Furthermore, the MRIO analysis results were compared with only one real SC.	As MRIO analysis was ruled out to support social hotspot assessment, another approach was needed.	Future research could try to produce a consistent, disaggregated synthesis of all MRIO databases. The comparison of MRIO analysis with real SC should be based on a larger sample of case studies.	By resorting to SOCLA and the support of emergent data collection providers, an efficient approach to the operationalization of S-LCA could be pursued.
Material-based risk assessment	The material-based risk assessment approach made it possible to calculate a material risk score and subsequently quantify the social impacts at component level	The rationale of the material-based social hotspot assessment is independent of products. Depending on the product of interest, the scope would have to be expanded to cover relevant materials.	The applied approach was a risk assessment rather than a social impact assessment. Additional production steps necessary to produce a component were not considered in the risk assessment. This might lead to crucial risks being overlooked.	In this thesis, only metals have been evaluated.	The expansion to other crucial production steps, like the manufacturing of a component, in the supply chain of components would improve the social risk assessment.	A possible expansion of the assessment approach could try to include the next group of relevant materials for the automotive industry, e.g. plastics. The implementation of SOLCA could be investigated.

	Key findings & added value	**Transferability of results**	**Methodological limitation & challenges**	**Practical limitation & challenges**	**Methodological steps forward**	**Practical steps forward**
Introductio of weights	Weighting appeared to more strongly influence by personal preference than by business function. Through the introduction of weights (35% for LCA, 34% for LCC and 31% for S-LCA) the calculation of an overall LCSA was possible. This enabled the evaluation of the effectiveness of product sustainability improvement measures.	The explicit weights in this thesis for the LCSA dimensions cannot be transferred to other companies or products as they reflect the trade-off decision process within the investigated company. The approach in this thesis — using Limit Conjoint Analysis — can be applied in any industry to elicit weights for the LCSA dimensions	In the study on weighting, only trade-offs between LCSA dimensions were evaluated, the indicators within each dimension were weighted equally. The evaluation of the ranking in the Limit Conjoint Analysis assumes distances between ranks to be equal. It could therefore not be captured, whether respondents saw large or small differences between certain stimuli.	The introduction of weights might be seen as critical as it might lead to LCSA practitioners presenting decision makers only with the aggregated results, while it is an important feature to still provide the underlying disaggregated results.	Future research could look at deriving the weights from other stakeholders, especially consumers would be of interest. The weighting of indicators within sustainability dimensions should also be looked at. Furthermore, the drivers that influence personal weighting of sustainability dimensions would be an interesting field of research.	Future research could test how well the results on an aggregated and disaggregated level are understood by decision makers.
Product Sustainability Budget	The introduction of the PSB made the implementation of improvement measures in product-related decision processes possible. To the author's knowledge, there exists no comparable approach.	The results of the specific study cannot be transferred as they have been derived from a specific study setup and sample. The type of study and he mechanism of the PSB could be applied anywhere. Results would strongly depend on the study setup and the kind of product sustainability features that as well as industry sector.	The WTP analysis was done for the German market. Also the WTP study quantified a hypothetical amount that consumers would be willing to pay.	The PSB has not been tested yet. Even though this research put forth strong evidence for a business case, the reality might look different.	An investigation of cost models could yield additional insights. Future research could look at how the impacts of a production process is affecting the local environment and how this, in turn, could affect the cost of the production process (e.g. excessive water consumption of a process leading to higher water costs).	The creation of a sustainability interior package and analysis of its real market and take rate data will determine the basis for an actual implementation of a PSB. A rather different approach to monetization and weighting of sustainability dimensions could also disregard all study results and rely on a strategic top-down decision.
Combinatio of Results	The developed approach was able — in the given context of an automotive company — to fulfill the initially set goal of this thesis. Data acquisition for S-LCA still remains a challenge.	The methods applied to create the operationalized LCSA framework that are independent of the company, e.g. the criticality scores, the Limit Conjoint Analysis etc., can be applied to any product. The specific results of this thesis, e.g. the weights for the LCSA dimensions, the amount of WTP for a sustainable interior package etc., cannot be transferred to other products.	The research objectives of this thesis were determined by the results of a structured literature review. Possible shortcomings in that analysis might have led to blind spots in the research needs. Therefore, the operationalized LCSA framework might be missing relevant aspects.	One of the challenges for the LCSA practitioner regarding the execution of the presented approach is to coordinate the expertise and responsibility of different departments within a company for the assessment of the impacts the different LCSA dimensions.	Future testing and practical experience from the application of this framework as well as monitoring the state of the art of the LCSA framework would be recommended.	Future testing and practical experience from the application of this framework as well as monitoring the state of the art of the LCSA framework would be recommended.

4.1 Added Scientific Value

This dissertation added scientific value by advancing the methodological development of LCSA, particularly at an automotive company. The main contributions were:

1. Structured identification of research needs to put LCSA into practice
2. Evaluation of an approach for assessing social hotspots via MRIO analysis
3. Development of an alternative, material-based approach to social hotspot assessment
4. Identification of a weighting set of sustainability dimensions to support decision making at an automotive company
5. Introduction of the Product Sustainability Budget to realize product sustainability improvement measures at a company despite rising costs
6. Combination of the individual findings to put the LCSA framework into practice at an automotive company enabling the identification and realization of product sustainability improvement potential

Also, the idea to break the LCSA down to the component level of a vehicle and express the relative sustainability impacts as criticality points has — the author's knowledge — not been presented before.

4.2 Recommendations for Future Research

The recommendations for future research are divided into recommendations for possible methodological steps forward (section 4.2.1) and practical steps forward (section 4.2.2).

4.2.1 Methodological Steps Forward

Future testing and practical experience from the application of this framework as well as monitoring the state of the art of the LCSA framework would be recommended. This would show, whether additional gaps will have to be addressed to identify and implement product sustainability improvement potential at an automotive company that have that have initially not been identified by the structured literature review. The granularity of assessment introduced in this thesis was the component level of vehicles. Even though this approach corresponded best with the product development process within an automotive company, this way of approaching the assessment also leads to measures that only have an effect at that level of granularity. This means that even though the greatest improvement potential at component level can be identified and implemented, a larger, because cross-cutting, lever could be overlooked. Therefore, another approach to identify the greatest improvement potential might be to look at cross-cutting issues like the materials used in the entire vehicle. This might — at least for the impact in the production phase — lead to a concentrated improvement approach that affects several components at once. A result might be a more effective product sustainability improvement than when targeting the component level. However, the cross-cutting measures are often managed independently from the product development cycles and are thus not (easily) addressed within product-related decision processes. Furthermore, the relation to the use phase impacts might be harder to determine for cross-cutting issues. It would still be an interesting idea to implement the assessment of such issues, e.g. the overall material use, in the existing framework as this analysis could be carried out complementary to the component level assessment. As the material composition of the components is known, the summation across all components can give the material composition of the entire vehicle. Table 11 shows how this additional consideration of the material use across the entire vehicle could be integrated into the presented LCSA framework. The proposal would be to sum the respective material across all components (see the columns two, three and four in Table 11). In analogy, the impacts on the respective LCSA dimensions that are related to the material production can also be determined across all components. Subsequently, the relative evaluation as applied to the component level could be applied to the material composition of the vehicle (see the last four rows in Table 11). Further research could look at implementing this additional aspect in the assessment.

Table 11: Possible expansion of the LCSA framework to evaluate the cross-cutting improvement potential of vehicle-wide material use

Component	**Quantity Steel**	**Quantity Aluminum**	**Quantity Copper**	**...**	**LCA Criticality Points**	**LCC Criticality Points**	**S-LCA Criticality Points**	**Overall LCSA Criticality Points**
A	8 kg	5 kg	2 kg	...	72	12	100	**59**
B	0 kg	3 kg	1 kg	...	82	17	57	**50**
C	2 kg	0 kg	0 kg	...	19	53	0	**26**
...	...	...	...	...	...	...	...	**...**
Entire vehicle	**1,500 kg**	**300 kg**	**80 kg**	**...**				
LCA Criticality Points	62	83	75	...				
LCC Criticality Points	53	72	80	...				
S-LCA Criticality Points	81	79	45	...				
Overall LCSA Criticality Points	**65**	**78**	**67**	...				

Regarding the material-based social risk assessment a possible expansion could try to include the next group of relevant materials for the automotive industry. Also, the expansion to other crucial production steps, like the manufacturing of a component, in the supply chain of components would improve the social risk assessment. Another expansion of the proposed LCSA framework worth pursuing would be the integration of the assessment of cross-cutting issues like the vehicle-wide material use. An approach to gather this information could be to identify the major plastics manufacturer and retrieve their main production sites. The next steps would follow the logic as presented in section 2.1.3, i.e. combined with the SHDB risk assessment for the countries of the production sites.

It would be an interesting next step to take the presented LCSA framework further to enable the monitoring of a component's/vehicle's LCSA impacts against a set target. This approach could then enable practitioners to also monitor and steer product sustainability improvement measures towards absolute target values for vehicles. Especially in light of possible absolute targets for GWP to stay within the 2°C target of the Paris Agreement (Gao et al., 2017), this would be an integral tool for companies.

The results from the proposed LCSA are based on the allocation of the use phase impacts to the component level of a vehicle via the mass of components. To mitigate the possible distortion introduced by this allocation approach, further research should investigate the relationship between components and their sustainability impacts in the use phase of a vehicle. This could be done, by first developing a way to determine the impact of a component on the fuel consumption in the use phase. Furthermore, the role of components in the aftersales market and warranty cases should be determined. Also, the social impacts of components in the use phase would be of interest.

The robustness of the set of the selected indicators could be investigated. This could be done by following the same approach as shown in section 2.1.1 and choosing a higher and lower threshold value, respectively, for the importance within the materiality matrix that determines inclusion or exclusion of a material topic. Karlewski stated that the application of S-LCA in the automotive industry still has quite some challenges to overcome and proposes the adoption of Social Organizational LCA (SOLCA) as viable option for this (Karlewski, 2016). This could be an approach worth pursuing further at an automotive company. The consideration of the assessment of possible positive social impacts would also be an option for further development of the method. Furthermore, the findings of this thesis regarding the assessment of the S-CLA dimension could be considered in the current efforts of updating the "Guidelines for Social Life Cycle Assessment of Products". The data basis for weighting could be expanded to different companies on the one hand and real consumers on the other hand. Future research could look at deriving the weights from other stakeholders than decision makers at automotive companies as well.

Also, the investigation of the link of sustainability impacts and future costs or avoided cost models (de Groot et al., 2002; de Groot, 2006) for the company that caused them would be of interest. It is conceivable that a detailed analysis of externalities might lead to an insight into how much a company indirectly has to pay for these externalities. A research could, for instance, look at how the water consumption of a production process in a region is affecting the local water supply. If it is leading to an increase in water prices in the long run, this would in turn affect the cost of the production process. Therefore, the risk of a future cost increase due to a current impact could be determined. Other possible impacts might be the use of fossil resources and the effects of their possible scarcity. Or the negative impact when health and safety requirements are not met which might lead to an increase in accidents and therefore an increase in costs. Future research could investigate this link between sustainability impacts and costs that indirectly arise to companies from them. The integration of such a kind of cost approach would render the weighting step of this thesis obsolete.

4.2.2 Practical Steps Forward

Regarding the next steps forward from a practitioner's point of view, the full application of LCSA framework under real conditions would be of interest. As no sustainability interior package has been realized as of yet, this is a potential for further research. Bringing about real market and take rate data will set a strong basis for an actual implementation of a PSB. If individual consumers fall behind expectations, corporate and institutional customers might still be a viable consumer group as green public procurement is more strongly supported in the EU (Commission of the European Communities, 2008). A business case for a sustainability interior package could be set up and the resulting PSB be applied at a company to test the acceptance and willingness to divert parts of revenue back to improving product sustainability. After application of the operationalized framework, it would be interesting to see, whether the identified top components and their improvement potential actually hold up, i.e. that the reason for their high overall sustainability impact can be identified and improved. Regarding the identification of social topics and indicators relevant to the company, this thesis relied on data from 2014. Next steps could look at, whether the identified social topics and indicators would still be relevant when considering more recent data. To address the challenge of data acquisition at a practical level, there is an increasing number of providers of software solutions that aim at facilitating the collection of data from value chain actors[13]. By resorting to SOCLA and the emergent data collection support providers, an efficient approach to the operationalization of S-LCA could be pursued. Next steps in terms of determining weighting sets would be to conduct the same study at other companies and with consumers. Furthermore, the same respondents should be asked

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[13] Examples for such providers would be SupplyShift, Stacksdata or Worldfavor.

again to find out how robust their initial weight distributions were. If a clear consumer expectation can be determined, that weighting set should be adopted in the long run after the currently proposed weighting set has gained acceptance within the operationalized LCSA framework. Also, when applying and testing the framework, it can be investigated, how aggregated and disaggregated results should best be presented to decision makers.

A rather different approach to monetization and weighting of sustainability dimensions could also disregard all study results and rely on a strategic top-down decision. For example, if a company sets up a rigorous product sustainability strategy that focuses on fostering environmental sustainability, it would be conceivable that this strategy also defines how the sustainability dimensions have to be weighed against each other, and be integrated into product-related decision making.

5 References

Arcese, G., Lucchetti, M. C., Massa, I., & Valente, C. (2018). State of the art in S-LCA: integrating literature review and automatic text analysis. *The International Journal of Life Cycle Assessment*, *23*(3), 394–405.

Azevedo, S. & Barros, M. (2017). The application of the triple bottom line approach to sustainability assessment: The case study of the UK automotive supply chain. *Journal of Industrial Engineering and Management*, *10*(2), 286.

Bach, V., Minkov, N., & Finkbeiner, M. (2018). Assessing the Ability of the Cradle to Cradle Certified$^{\text{TM}}$ Products Program to Reliably Determine the Environmental Performance of Products. *Sustainability*, *10*(5), 1562.

Bare, J. C., Hofstetter, P., Pennington, D. W., & de Haes, H. A. U. (2000). Midpoints versus endpoints: The sacrifices and benefits. *The International Journal of Life Cycle Assessment*, *5*(6), 319–326.

Behzadian, M., Kazemzadeh, R., Albadvi, A., & Aghdasi, M. (2010). PROMETHEE: A comprehensive literature review on methodologies and applications. *European Journal of Operational Research*, *200*(1), 198–215.

Benayoun, R., Roy, B., & Sussmann, B. (1966). ELECTRE: une méthode pour guider le choix en présence de points de vue multiples SEMA. *Direction Scientifique, Notre de travail*, (49).

Benoît Norris, C. (2014). Data for social LCA.

Benoit-Norris, C., Cavan, D. A., & Norris, G. A. (2012). Identifying social impacts in product supply chains: Overview and application of the social hotspot database. *Sustainability*, *4*(9), 1946–1965.

Benoít Norris, C. & Mazijn, B. (2009). *Guidelines for social life cycle assessment of products*. UNEP/Earthprint.

Benoít Norris, C., Norris, G. A., & Aulisio Cavan, D. (2013). Social Hotspots Database - Supporting Documentation.

Biesalski & Company (2014). Wertschöfungsreport Nachhaltigkeit 2014 - Was ist gutes Gewissen wert?

BMW Group (2014). Sustainable Value Report.

Brans, J. P. (1982). L'ingénièrie de la décision; Elaboration d'instruments d'aide à la décision. La méthode PROMETHEE. In Nadeau, R. & Landry, M. (Eds.), *L'aide à la décision: Nature, Instruments et Perspectives d'Avenir*, (pp. 183–213)., Québec, Canada. Presses de l'Université Laval.

Brooker, A. D., Ward, J., & Wang, L. (2013). Lightweighting Impacts on Fuel Economy, Cost, and Component Losses.

Chan, J. & Pun, N. (2010). Suicide as protest for the new generation of Chinese migrant workers : Foxconn, global capital, and the state. *Asia-Pacific journal : Japan focus*, *8*(37), 1–33.

Ciravegna, L., Romano, P., & Pilkington, A. (2013). Outsourcing practices in automotive supply networks: an exploratory study of full service vehicle suppliers. *International Journal of Production Research*, *51*(8), 2478–2490.

Ciroth, A. & Eisfeldt, F. (2015). A new, comprehensive database for social LCA: PSILCA. Technical report, GreenDelta.

Commission of the European Communities (2008). Public procurement for a better environment.

Čuček, L., Klemeš, J. J., & Kravanja, Z. (2012). A Review of Footprint analysis tools for monitoring impacts on sustainability. *Journal of Cleaner Production*, *34*, 9–20.

Curran, R., Raghunathan, S., & Price, M. (2004). Review of aerospace engineering cost modelling: The genetic causal approach. *Progress in Aerospace Sciences*, *40*(8), 487–534.

Davies, G. R. (2013). Appraising weak and strong sustainability: searching for a middle ground. *Columbia University Academic Commons.*

de Groot, R. (2006). Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. *Landscape and Urban Planning*, *75*(3-4), 175–186.

de Groot, R. S., Wilson, M. A., & Boumans, R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, *41*(3), 393–408.

Dreyer, L., Hauschild, M., & Schierbeck, J. (2006). A Framework for Social Life Cycle Impact Assessment (10 pp). *The International Journal of Life Cycle Assessment*, *11*(2), 88–97.

Elkington, J. (1998). Partnerships from cannibals with forks: The triple bottom line of 21st-century business. *Environmental Quality Management*, *8*(1), 37–51.

Finkbeiner, M. (2014). Product environmental footprint - Breakthrough or breakdown for policy implementation of life cycle assessment? *International Journal of Life Cycle Assessment*, *19*(2), 266–271.

Finkbeiner, M., Inaba, A., Tan, R., Christiansen, K., & Klüppel, H.-J. (2006). The new international standards for life cycle assessment: ISO 14040 and ISO 14044. *The international journal of life cycle assessment*, *11*(2), 80–85.

Finkbeiner, M., Reimann, K., & Ackermann, R. (2008). Life cycle sustainability assessment (LCSA) for products and processes. In *SETAC Europe 18th Annual Meeting, 2529 May*, Warsaw, Poland.

Finkbeiner, M., Schau, E. M., Lehmann, A., & Traverso, M. (2010). Towards Life Cycle Sustainability Assessment. *Sustainability*, *2*(10), 3309–3322.

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., & Suh, S. (2009). Recent developments in Life Cycle Assessment. *Journal of Environmental Management*, *91*(1), 1–21.

Fontes, J., Bolhuis, A., Bogaers, K., Saling, P., van Gelder, R., Traverso, M., & Das Gupta, J. (2015). *Handbook for Product Social Impact Assessment* (2.0 ed.). Amsterdam: PRé Sustainability.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D., Haywood, J., Lean, J., Lowe, D., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., & Van Dorland, R. (2007). Changes in Atmospheric Constituents and in Radiative Forcing. In S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. Averyt, M.Tignor, & H. Miller (Eds.), *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 234). Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Freidberg, S. (2013). Calculating sustainability in supply chain capitalism. *Economy and Society*, 1–26.

Fritz, M. M. C., Schöggl, J. P., & Baumgartner, R. J. (2018). Enabling a Supply Chain-Wide Sustainability Assessment: A Focus on the Electronics and Automotive Industries. In *Social and Environmental Dimensions of Organizations and Supply Chains* (pp. 61–77). Cham: Springer International Publishing.

Gao, Y., Gao, X., & Zhang, X. (2017). The 2 °C Global Temperature Target and the Evolution of the Long-Term Goal of Addressing Climate Change – From the United Nations Framework Convention on Climate Change to the Paris Agreement. *Engineering*, *3*(2), 272–278.

Ghadimi, P., Azadnia, A. H., Mohd Yusof, N., & Mat Saman, M. Z. (2012). A weighted fuzzy approach for product sustainability assessment: a case study in automotive industry. *Journal of Cleaner Production*, *33*, 10–21.

Gibbs, D. C., Longhurst, J., & Braithwaite, C. (1998). 'Struggling with Sustainability': Weak and Strong Interpretations of Sustainable Development within Local Authority Policy. *Environment and Planning A*, *30*(8), 1351–1365.

Global Reporting Initiative (2015). G4 Sustainability Reporting Guidelines – Implementation Manual. Technical report.

Global Reporting Initiative (2018). Consolidated Set of GRI Sustainability Reporting Standards 2018. Technical report.

Gluch, P. & Baumann, H. (2004). The life cycle costing (LCC) approach: A conceptual discussion of its usefulness for environmental decision-making. *Building and Environment*, *39*(5), 571–580.

González, P., Sarkis, J., & Adenso Díaz, B. (2008). Environmental management system certification and its influence on corporate practices. *International Journal of Operations & Production Management*, *28*(11), 1021–1041.

Govindan, K. & Jepsen, M. B. (2016). ELECTRE: A comprehensive literature review on methodologies and applications. *European Journal of Operational Research*, *250*(1), 1–29.

Grießhammer, R., Buchert, M., Gensch, C., Hochfeld, C., Manhart, A., Rüdenauer, I., & Ebinger, F. (2007). *PROSA - Product Sustainability Assessment* (Beschreibu ed.).

Heijungs, R., Huppes, G., & Guinée, J. B. (2009). A scientific framework for LCA. Technical report, Leiden University (CML).

Heijungs, R., Settanni, E., & Guinée, J. (2013). Toward a computational structure for life cycle sustainability analysis: Unifying LCA and LCC. *International Journal of Life Cycle Assessment*, *18*(9), 1722–1733.

Hooftman, N., Messagie, M., Van Mierlo, J., & Coosemans, T. (2018). A review of the European passenger car regulations – Real driving emissions vs local air quality. *Renewable and Sustainable Energy Reviews*, *86*, 1–21.

Hunkeler, D., Lichtenvort, K., & Rebitzer, G. (2008). *Environmental Life Cycle Costing*. SETAC/CRC Press.

Hutchins, M. J. & Sutherland, J. W. (2008). An exploration of measures of social sustainability and their application to supply chain decisions. *Journal of Cleaner Production*, *16*(15), 1688–1698.

Iofrida, N., Strano, A., Gulisano, G., & De Luca, A. I. (2018). Why social life cycle assessment is struggling in development? *The International Journal of Life Cycle Assessment*, *23*(2), 201–203.

ISO 14025 (2006). Environmental labels and declarations – Type III environmental declarations – Principles and procedures.

ISO 14040 (2006). Environmental management – Life Cycle Assessment – Principles and Framework. *Environmental Management*, *3*, 28.

ISO 14044 (2006). Life cycle assessment – Requirements and guidelines.

ISO 26000 (2010). Guidance on social responsibility.

Jasinski, D., Meredith, J., & Kirwan, K. (2015). A comprehensive review of full cost accounting methods and their applicability to the automotive industry. *Journal of Cleaner Production.*

Jasiński, D., Meredith, J., & Kirwan, K. (2016). A comprehensive framework for automotive sustainability assessment. *Journal of Cleaner Production*, *135*, 1034–1044.

Jørgensen, A., Bocq, A. L., Nazarkina, L., & Hauschild, M. (2008). Methodologies for Social Life Cycle Assessment. *The International Journal of Life Cycle Assessment*, *13*(2), 96–103.

Jørgensen, A., Finkbeiner, M., Jørgensen, M. S., & Hauschild, M. Z. (2010). Defining the baseline in social life cycle assessment. *The International Journal of Life Cycle Assessment*, *15*(4), 376–384.

Jörissen, J., Kopfmüller, J., Brandl, V., & Paetau, M. (1999). *Ein integratives Konzept nachhaltiger Entwicklung.* Karlsruhe: Forschungszentrum Karlsruhe GmbH.

JRC (2012). Towards a Life-Cycle Based European Sustainability Footprint Framework. Technical report, European Commission - Joint Research Centre: Institute for Environment and Sustainability, Luxembourg.

Karaaslan, E., Zhao, Y., & Tatari, O. (2017). Comparative life cycle assessment of sport utility vehicles with different fuel options. *The International Journal of Life Cycle Assessment.*

Karlewski, H. (2016). *Social life cycle assessment in the automotive industry.* Phd thesis, TU Berlin.

Klöpffer, W. (2003). Life-Cycle based methods for sustainable product development. *The International Journal of Life Cycle Assessment*, *8*(3), 157–159.

Klöpffer, W. (2008). Life cycle sustainability assessment of products. *The International Journal of Life Cycle Assessment*, *13*(2), 89–95.

Klöpffer, W. & Grahl, B. (2014). From LCA to Sustainability Assessment. In *Life Cycle Assessment (LCA)* (pp. 357–374). Weinheim: Wiley-VCH Verlag GmbH & Co. KGaA.

Kölsch, D., Saling, P., Kicherer, A., Sommer, A. G., & Schmidt, I. (2008). How to measure social impacts? A socio-eco-efficiency analysis by the SEEBALANCE® method. *International Journal of Sustainable Development*, *11*(1), 1.

Koplin, J., Seuring, S., & Mesterharm, M. (2007). Incorporating sustainability into supply management in the automotive industry - the case of the Volkswagen AG. *Journal of Cleaner Production*, *15*(11–12), 1053–1062.

Kucukvar, M., Noori, M., Egilmez, G., & Tatari, O. (2014). Stochastic decision modeling for sustainable pavement designs. *The International Journal of Life Cycle Assessment*, *19*(6), 1185–1199.

Lehmann, A., Berger, M., & Finkbeiner, M. (2017). Life Cycle Based CO2 Emission Credits: Options for Improving the Efficiency and Effectiveness of Current Tailpipe Emissions Regulation in the Automotive Industry. *Journal of Industrial Ecology*.

Lehmann, A., Finkbeiner, M., Broadbent, C., & Balzer, R. (2015). Policy Options for Life Cycle Assessment Deployment in Legislation. In G. Sonnemann & M. Margni (Eds.), *LCA Compendium - The Complete World of Life Cycle Assessment* chapter 15, (pp. 213–224). Dordrecht Heidelberg New York London: Springer.

Lehmann, A., Zschieschang, E., Traverso, M., Finkbeiner, M., & Schebek, L. (2013). Social aspects for sustainability assessment of technologies – challenges for social life cycle assessment (SLCA). *The International Journal of Life Cycle Assessment*, *18*(8), 1581–1592.

List, J. A. & Gallet, C. A. (2001). What Experimental Protocol Influence Disparities Between Actual and Hypothetical Stated Values? *Environmental and Resource Economics*, *20*(3), 241–254.

Liu, J.-h. & Meng, Z. (2017). Innovation Model Analysis of New Energy Vehicles: Taking Toyota, Tesla and BYD as an Example. *Procedia Engineering*, *174*, 965–972.

Manzardo, A., Ren, J., Mazzi, A., & Scipioni, A. (2012). A grey-based group decision-making methodology for the selection of hydrogen technologies in life cycle sustainability perspective. *International Journal of Hydrogen Energy*, *37*(23), 17663–17670.

Martínez-Blanco, J., Lehmann, A., Chang, Y.-J., & Finkbeiner, M. (2015). Social organizational LCA (SOLCA) – a new approach for implementing social LCA. *The International Journal of Life Cycle Assessment*, *20*(11), 1586–1599.

Mayyas, A., Omar, M., Hayajneh, M., & Mayyas, A. R. (2017). Vehicle's Lightweight Design vs. Electrification from Life Cycle Assessment Perspective. *Journal of Cleaner Production*.

Mayyas, A., Qattawi, A., Omar, M., & Shan, D. (2012). Design for sustainability in automotive industry: A comprehensive review. *Renewable and Sustainable Energy Reviews*, *16*(4), 1845–1862.

Mazur, C., Contestabile, M., Offer, G. J., & Brandon, N. (2015). Understanding the drivers of fleet emission reduction activities of the German car manufacturers. *Environmental Innovation and Societal Transitions*, *16*, 3–21.

Meadows, D., Meadows, D., Randers, J., & Behrens, W. (1972). *The Limits to Growth*, volume 205. Universe Books.

Meadows, D., Randers, J., & Meadows, D. (2007). *Limits to Growth – the 30 Year Update*. London: Earthscan.

Miemczyk, J., Johnsen, T. E., & Macquet, M. (2012). Sustainable purchasing and supply management: a structured literature review of definitions and measures at the dyad, chain and network levels. *Supply Chain Management: An International Journal*, *17*(5), 478–496.

Minkov, N., Bach, V., & Finkbeiner, M. (2018). Characterization of the Cradle to Cradle Certified$^{\mathrm{TM}}$ Products Program in the Context of Ecolabels and Environmental Declarations. *Sustainability*, *10*(3), 738.

Moran, D., McBain, D., Kanemoto, K., Lenzen, M., & Geschke, A. (2015). Global Supply Chains of Coltan. *Journal of Industrial Ecology*, *19*(3), 357–365.

Murphy, J. J., Allen, P. G., Stevens, T. H., & Weatherhead, D. (2005). A Meta-analysis of Hypothetical Bias in Stated Preference Valuation. *Environmental & Resource Economics*, *30*(3), 313–325.

Ness, B., Urbel-Piirsalu, E., Anderberg, S., & Olsson, L. (2007). Categorising tools for sustainability assessment. *Ecological Economics*, *60*(3), 498–508.

Neugebauer, S., Forin, S., & Finkbeiner, M. (2016). From Life Cycle Costing to Economic Life Cycle Assessment – Introducing an Economic Impact Pathway. *Sustainability*, *8*(5), 428.

Neugebauer, S., Martinez-Blanco, J., Scheumann, R., & Finkbeiner, M. (2015). Enhancing the practical implementation of life cycle sustainability assessment – proposal of a Tiered approach. *Journal of Cleaner Production*, *102*, 165–176.

OECD (2012). OECD Due Diligence Guidance for Responsible Supply Chains of Minerals from Conflict-Affected and High-Risk Areas: Third Edition.

Onat, N., Kucukvar, M., & Tatari, O. (2014). Towards Life Cycle Sustainability Assessment of Alternative Passenger Vehicles. *Sustainability*, *6*(12), 9305–9342.

Opricovic, S. & Tzeng, G.-H. (2004). Compromise solution by MCDM methods: A comparative analysis of VIKOR and TOPSIS. *European Journal of Operational Research*, *156*(2), 445–455.

Opricovic, S. & Tzeng, G.-H. (2007). Extended VIKOR method in comparison with outranking methods. *European Journal of Operational Research*, *178*(2), 514–529.

Parent, J., Cucuzzella, C., & Revéret, J.-P. (2012). Revisiting the role of LCA and SLCA in the transition towards sustainable production and consumption. *The International Journal of Life Cycle Assessment*, *18*(9), 1642–1652.

Petti, L., Serreli, M., & Di Cesare, S. (2018). Systematic literature review in social life cycle assessment. *The International Journal of Life Cycle Assessment*, *23*(3), 422–431.

Pizzol, M., Weidema, B., Brandão, M., & Osset, P. (2015). Monetary valuation in Life Cycle Assessment: a review. *Journal of Cleaner Production*, *86*, 170–179.

Rebitzer, G., Hunkeler, D., & Jolliet, O. (2003). LCC-The economic pillar of sustainability: Methodology and application to wastewater treatment. *Environmental Progress*, *22*(4), 241–249.

Redelbach, M., Klötzke, M., & Friedrich, H. E. (2012). Impact of lightweight design on energy consumption and cost effectiveness of alternative powertrain concepts. In *EEVC*, Brussels.

Ren, J., Manzardo, A., Mazzi, A., Zuliani, F., & Scipioni, A. (2015). Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multicriteria decision-making. *The International Journal of Life Cycle Assessment*, *20*(6), 842–853.

Report Of The United Nations Conference On And Development (1992). Rio Declaration On Environment And Development. Technical report.

Reynolds, C. & Kandlikar, M. (2007). How hybrid-electric vehicles are different from conventional vehicles: the effect of weight and power on fuel consumption. *Environmental Research Letters*, *2*(1), 014003.

Rodrigues Vaz, C., Shoeninger Rauen, T. R., & Rojas Lezana, Á. G. (2017). Sustainability and Innovation in the Automotive Sector: A Structured Content Analysis. *Sustainability*, *9*(6), 880.

Saaty, T. L. (1990). How to make a decision: The analytic hierarchy process. *European Journal of Operational Research*, *48*(1), 9–26.

Sala, S., Vasta, A., Mancini, L., Dewulf, J., & Rosenbaum, E. (2015). Social Life Cycle Assessment - State of the art and challenges for supporting product policies. Technical report, JRC Technical Reports.

Saling, P., Kicherer, A., Dittrich-Krämer, B., Wittlinger, R., Zombik, W., Schmidt, I., Schrott, W., & Schmidt, S. (2002). Eco-efficiency analysis by basf: the method. *The International Journal of Life Cycle Assessment*, *7*(4), 203–218.

Saling, P., Maisch, R., Silvani, M., & König, N. (2005). Assessing the Environmental-Hazard Potential for Life Cycle Assessment, Eco-Efficiency and SEEbalance (8 pp). *The International Journal of Life Cycle Assessment*, *10*(5), 364–371.

Schmidt, W.-P. & Taylor, A. (2006). Ford of Europe's product sustainability index. In *Proceedings of 13th CIRP International Conference on Life Cycle Engineering. Leuven May 31st–June 2nd*, (pp. 5–10). Citeseer.

Seghezzo, L. (2009). The five dimensions of sustainability. *Environmental Politics*, *18*(4), 539–556.

Simão, L. & Lisboa, A. (2017). Green Marketing and Green Brand - The Toyota Case. *Procedia Manufacturing*, *12*, 183–194.

Singh, R. K., Murty, H. R., Gupta, S. K., & Dikshit, A. K. (2009). An overview of sustainability assessment methodologies. *Ecological Indicators*, *15*(1), 281–299.

Spangenberg, J., Pfahl, S., & Deller, K. (2002). Towards indicators for institutional sustainability: lessons from an analysis of Agenda 21. *Ecological Indicators*, *2*(1-2), 61–77.

Srivastava, S. K. (2007). Green supply-chain management: A state-of-the-art literature review. *International Journal of Management Reviews*, *9*(1), 53–80.

Stone, F. (2004). Deconstructing silos and supporting collaboration. *Employment Relations Today*, *31*(1), 11–18.

Sukitsch, M., Engert, S., & Baumgartner, R. (2015). The Implementation of Corporate Sustainability in the European Automotive Industry: An Analysis of Sustainability Reports. *Sustainability*, *7*(9), 11504–11531.

Swarr, T. E., Hunkeler, D., Klöpffer, W., Pesonen, H. L., Ciroth, A., Brent, A. C., & Pagan, R. (2011). Environmental life-cycle costing: A code of practice. *International Journal of Life Cycle Assessment*, *16*(5), 389–391.

Tarne, P., Lehmann, A., & Finkbeiner, M. (2018a). A comparison of Multi-Regional Input-Output databases regarding transaction structure and supply chain analysis. *Journal of Cleaner Production*, *196*, 1486–1500.

Tarne, P., Lehmann, A., & Finkbeiner, M. (2018b). Introducing weights to life cycle sustainability assessment – how do decision-makers weight sustainability dimensions? *The International Journal of Life Cycle Assessment*.

Tarne, P., Lehmann, A., Kantner, M., & Finkbeiner, M. (2018). Introducing a product sustainability budget at an automotive company – one option to increase the use of LCSA results in decision-making processes. *The International Journal of Life Cycle Assessment*.

Tarne, P., Traverso, M., & Finkbeiner, M. (2017). Review of Life Cycle Sustainability Assessment and Potential for its Adoption at an Automotive Company. *Sustainability*, *9*(4), 670.

Traverso, M. (2018). Is social life cycle assessment really struggling in development or is it on a normal path towards harmonization/standardization? *The International Journal of Life Cycle Assessment*, *23*(2), 199–200.

Traverso, M., Finkbeiner, M., Jørgensen, A., & Schneider, L. (2012). Life Cycle Sustainability Dashboard. *Journal of Industrial Ecology*, *16*(5), 680–688.

Tukker, A., Huppes, G., Guinée, J., Heijungs, R., de Koning, A., Van Oers, L., Suh, S., Geerken, T., Van Holderbeke, M., Jansen, B., & Nielsen, P. (2006). Environmental Impact of Products (EIPRO) - Analysis of the life cycle environmental impacts related to the final consumption of the EU-25. *Analysis*, *22284*(May), 1–13.

Tukker, A. & Jansen, B. (2006). Environmental Impacts of Products: A Detailed Review of Studies. *Journal of Industrial Ecology*, *10*(3), 159–182.

UN (2011). Guiding Principles on Business and Human Rights. Technical report, United Nations Human Rights.

UNEP/SETAC (2013). The Methodological Sheets for Subcategories in Social Life Cycle Assessment (S-LCA).

UNEP/SETAC Life Cycle Initiative (2011). *Towards a Life Cycle Sustainability Assessment - Making informed choices on products*. UNEP/SETAC Life Cycle Initiative.

United States Geological Survey (2013). National Minerals Information Center.

Verisk Maplecroft (2017). Verisk Maplecroft.

Wang, X. & Chan, H. K. (2013). An integrated fuzzy approach for evaluating remanufacturing alternatives of a product design. *Journal of Remanufacturing*, *3*(1), 10.

WCED (1987). *Our Common Future*, volume Report of.

Weymar, E. & Finkbeiner, M. (2016). Statistical analysis of empirical lifetime mileage data for automotive LCA. *The International Journal of Life Cycle Assessment*, *21*(2), 215–223.

Whitehead, J. (2017). Prioritizing Sustainability Indicators: Using Materiality Analysis to Guide Sustainability Assessment and Strategy. *Business Strategy and the Environment*, *26*(3), 399–412.

Wood, R. & Hertwich, E. G. (2013). Economic modelling and indicators in life cycle sustainability assessment. *The International Journal of Life Cycle Assessment*, *18*(9), 1710–1721.

Zamagni, A. (2012). Life cycle sustainability assessment. *The International Journal of Life Cycle Assessment*, *17*(4), 373–376.