Sustainable Production, Life Cycle Engineering and Management

Series Editors: Christoph Herrmann, Sami Kara

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Progress in Life Cycle Assessment 2018

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Series editors

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Modern production enables a high standard of living worldwide through products and services. Global responsibility requires a comprehensive integration of sustainable development fostered by new paradigms, innovative technologies, methods and tools as well as business models. Minimizing material and energy usage, adapting material and energy flows to better fit natural process capacities, and changing consumption behaviour are important aspects of future production. A life cycle perspective and an integrated economic, ecological and social evaluation are essential requirements in management and engineering. This series will focus on the issues and latest developments towards sustainability in production based on life cycle thinking.

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Progress in Life Cycle Assessment 2018

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Foreword

The principles of sustainable development, that meet the needs of present and future generations, have been introduced to corporate management already in the 1990s. These days, especially resource efficiency throughout supply chains is one of the most striking challenges for companies. Expertise in the field of life cycle assessment (LCA) is a key competence to meet it.

LCA methodology already proved its ability to measure environmental impacts of complex systems in various disciplines, application scenarios, and industries. Thereby, LCA addresses a huge number of concepts. Since the 1970s, LCA has been continuously extended and developed further in order to account for new insights and methodological developments and to reflect new application areas arising through technological innovations. However, when trying to adopt LCA methods to specific application scenarios, researchers and practitioners still face problems as to comparability (e.g., for benchmarking issues), recognition, credibility, and transparency of LCA. This leads to research questions: Why did earlier applications of LCA not lead to a broader diffusion of the methodology into management and research? How can the impact assessment results be interpreted in order to better support decision making in companies? How can LCA methodologies be adjusted to better meet the characteristics of management and various industries and research disciplines?

In view of this situation, LCA research is challenged to enlarge its focus from environmental issues to the broader concept of corporate sustainability with its three pillars: economic, social, and environmental. LCA also could contribute to a paradigm shift which leads to the development and establishment of strategic sustainable management in order to support the strategic goal of sustainable development in companies. The data derived by LCA could also be further processed (e.g., by decision analysis, business analytics) in order to better support decision making. Therefore, the interpretation phase of LCA might also be adjusted for a better communication of the LCA results to internal or external stakeholders.

This volume includes contributions by researchers from various disciplines, such as industrial ecology, biotechnology, information systems research, agriculture and energy management, inter alia. It impressively addresses the mentioned issues above and shows that LCA can be applied for decision making in various disciplines, on various levels and within a variety of organizations, ranging from material comparisons and technology assessments at the corporate level to meso- and macro-level studies assessing the impact of environmental policies. The included chapters, all based on the scientific status quo in LCA methodology, underline the importance of the definition of goal and scope of the LCA, as a presetting of the selected methodologies, as well as the choice of data. Especially, the choice of a functional unit and system boundaries to define the scope of the LCA has a significant influence on comparability, recognition, credibility, and transparency of LCA results.

This volume demonstrates impressively how the potential of LCA can be used in various viable application scenarios. It is a means of orientation and hopefully has a broad impact on both researchers and practitioners in the near future. I hope that the scientific community and practitioners will draw manifold inspirations for their own work from this book and that this book may encourage researchers and companies to make use of LCA in order to contribute to the long-term goal of sustainable development.

Oldenburg, Germany
December 2018

Prof. Dr.-Ing. habil. Jorge Marx Gómez

Preface

The implications of environmental damage, climate change, and resource depletion for human well-being, other species, and the planet as a whole have been at the center of interest for several decades. In order to reach the goals set for sustainable development by the United Nations, European Union, and other supranational organizations, comprehensive and robust tools are required that support decision makers in identifying those solutions that best support the desired sustainable development goals. Life cycle assessment (LCA) represents such a tool, as it is used to assess the impacts of product and service systems across their entire life cycle, from raw material extraction to the end-of-life stage. Considering all relevant impacts caused by a potential solution across its life cycle and thus adopting a system perspective, LCA can be applied for decision making in various disciplines, on various levels and within a variety of organizations, ranging from material comparisons and technology assessments at the corporate-level to macro-level studies assessing the impact of environmental policies.

Since its inception in the 1970s, LCA has been continuously extended and developed further in order to account for new insights and methodological developments and to reflect new areas of application arising through technological innovations. It is in this context that the Ökobilanzwerkstatt was established in 2005 as a forum for junior researchers in the field of life cycle assessment. Since then, the annually held Ökobilanzwerkstatt has provided junior researchers focusing on fundamental LCA methodology or applying LCA in various disciplines with an opportunity for scientific discussion, exchange of experiences, and advancement of methodology.

This volume includes contributions by researchers from various disciplines, such as industrial ecology, biotechnology, information systems research, agriculture and energy management, inter alia, who have participated in the 14th Ökobilanzwerkstatt held at the Deutsche Bundesstiftung Umwelt in Osnabrück in October 2018.

Part I of this book provides an introduction to methodological developments in the research area of life cycle assessment. In their publication, Pohl et al. emphasize the relevance of user decision and behavior in LCA and discuss related modeling aspects with regard to the definition of system boundaries, the definition of the use phase, and the collection of inventory data. Brinkmann and Metzger, on the other hand, explain the ecological single-score method based on ecological product declaration data and validate it against common single-score assessments (ReCiPe and UBP). Showing five main categories of interest in social sustainability, the contribution by Hösel et al. analyzes currently effective frameworks, process guidelines, and management approaches in terms of social sustainability criteria.

Part II encompasses three contributions that are concerned with the application of the topic of LCA to the field of mobility. The article by Neef et al. determines common characteristics of studies comparing life cycle carbon emissions of mobility services and passenger vehicles and shows that current life cycle assessment (LCA)-based approaches mostly apply the two methodological characteristics: (1) person-km (p-km) is used as reference unit to compare carbon performances across transport modes and (2) scenario analyses are used to deal with the poor data basis and disruptive character of mobility services. In preparation of an in-depth technology assessment, Wittstock and Teuteberg present a scoping study aimed at achieving a better conceptualization of what core elements constitute mobility as a service, what risks and opportunities are associated with this concept, and how these may be further analyzed as part of a technology assessment project. A dynamization and modularization of the classic LCA approach is proposed by Pichlmaier et al. in order to easily integrate the simulated electricity generation from energy system models on an hourly basis as well as future energy technologies. A special focus is put on Power-to-X (PtX) technologies in the transport sector due to its potential in deep decarbonization scenarios.

The two chapters of **Part III** are concerned with the application of the LCA methodology to the field of energy management. To investigate the applicability of external costs for the environmental assessment of power systems, Lazar and Tietze integrate external costs into the method of life cycle assessment on the case of power generation technologies. The correlation between the LCA results considering external costs on the one hand and on the other hand standard midpoint impact assessment is investigated by regression analysis. Mühlbach et al., on the other hand, develop a life cycle assessment tool that combines the embodied energy (energy used for the production of a building) as well as the energy consumption of existing buildings. By combining LCA data with a customized extract from the 2011 census for parts for Lower Saxony, the tool allows for spatially explicit assessments on a square kilometer grid.

Finally, **Part IV** combines three contributions discussing the application of LCA to the fields of production and logistics, respectively. Bussa et al. apply a life cycle assessment approach in order to evaluate the potential of cyanobacterial biomass as a replacement of maize as feedstock for polylactic acid, to identify the drivers of the environmental impacts and to assess three different improvement scenarios. Assessing the sustainability of dairy farms in Central Germany, Heider-van-Diepen demonstrates the application of “REPRO,” a software which makes it possible to represent agricultural actions and consequences. The climate impact is illustrated by a greenhouse gas balance, the ratio of nutrient emissions and energy intensity in their carbon dioxide equivalents to the product. Finally, from the perspective of a logistics provider, Hülemeyer and Schoeder compare two existing approaches for carbon accounting, the European standard EN 16258 and the GLEC framework.

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Frank Teuteberg
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First and foremost, the editors would like to thank all authors for their excellent contributions to this volume and for shouldering the bulk of the efforts to realize this project. The high quality of all paper included here was assured by a blind review process. Furthermore, we would like to thank Springer Publishing, especially Silvia Schilgerius and Ramesh Kumaran, who have supported this project at all phases of development in a constructive, professional, friendly, and open manner. This book could not have been completed without professional preparation and organization. Therefore, we would particularly like to thank Rikka Wittstock, University of Osnabruck, for her assistance in coordination, preparation, and communication, as well as proofreading services. We hope that this book will receive widespread recognition both from practitioners and the scientific community. As for the contents of this volume, the editors are always open for any suggestions for improvement and are looking forward to the readers' comments and fruitful discussions.

Osnabrück/Darmstadt, Germany
December 2018

Frank Teuteberg
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About the Editors

Prof. Dr. Frank Teuteberg has been heading the Department of Accounting and Information Systems at the Osnabrück University since 2007. He received five prestigious professorship offers (W3) [(Rostock (2007), Osnabrück (2007), Innsbruck (2008), Chemnitz (2011), Osnabrück (2011)]. Further, he acts as Reviewer of applications for third-party funding for the German Research Foundation (DFG), the German Federal Environmental Foundation (DBU) as well as the National Research Foundation, South Africa (NRF). Also, he is Member of the "Assessment Committee" of the Netherlands' Organization for Scientific Research (NWO), which is under the auspices of the Dutch Ministry of Infrastructure and Environment. He has authored more than 300 scientific publications in leading journals and conferences (A and B ranking according to VHB Jourqual3) inter alia in the relevant research areas of eHealth, cloud computing, blockchain, sustainable supply chain management, and sustainability reporting. In the current CHE ranking, the field of information systems at Osnabrück University ranks among the German top group in the category research (publications). Furthermore, he is spokesman of the profile line "Digital Society - Innovation - Regulation" at the Osnabrück University.

Dr. Maximilian Hempel received his Ph.D. at the Technical University of Hamburg-Harburg on Mercury in the Environment. From 1990 to 1996, he worked at the Institute for Chemistry of the Helmholtz Centre Geesthacht. Since 2002, he is working for the German Federal Environmental Foundation (DBU). He is in charge of the Department of Environmental Chemistry and the project group Resource Efficiency. His main fields of interest are green and sustainable chemistry and pharmacy, resource efficiency as well as innovation management. He is Member of the Board of the working group "Sustainable Chemistry" within the German Chemical Society (GDCh) and Member of the Environmental Board for the Federal Authority for Safety in Technology and Chemistry (BAM, Germany). He edited 5 books and published more than 30 peer-reviewed papers.

Prof. Liselotte Schebek heads the Department of Material Flow Management and Resource Management at the Faculty of Civil and Environmental Engineering and is also Scientific Director of the "Secondary Raw Materials" Division of the Fraunhofer Project Group Materials Recycling and Resource Strategies IWKS. Her research interests lie at the interface of environmental, construction, and infrastructure research in the engineering sciences with systems analytical modeling, with a focus on material flow analyses, life cycle assessment, and scenario techniques. She leads numerous research projects on the analysis of environmental and sustainability aspects of new technologies, the evaluation of energy and resource efficiency with the development of indicators, and environmental services. She is a member of the Energy Science and Engineering Graduate School, in which she coordinates the Cross-Sectional Research Platform "Assessment Methods" and participates in the Technology Platform "Building Integration and Energy Self-sustaining Settlement."

Part I
Methodological Developments

Beyond Production—the Relevance of User Decision and Behaviour in LCA

Johanna Pohl, Paul Suski, Franziska Haucke, Felix M. Piontek and Michael Jäger

Abstract The way in which products and services are used can have a significant impact on their environmental performance. Practice shows, however, that life cycle assessment (LCA) studies often either assume average usage parameters, or only address a limited number of life cycle phases (‘cradle to gate’), without considering the use phase. This chapter therefore aims to emphasize the relevance of user decision and behaviour in LCA and to discuss related modelling aspects with regard to the definition of system boundaries, the definition of the use phase and the collection of inventory data. Furthermore, processes of decision-making in the context of LCA are critically reflected and suggestions for improvements are discussed.

Keywords Life cycle assessment (LCA) · Product-service system (PSS) · Sharing economy · Use phase modelling · Rebound effect · Decision making

1 Introduction

User decisions can have far-reaching effects on the environmental performance of products and services [1–4]. Therefore, when assessing the life-cycle-wide envi-

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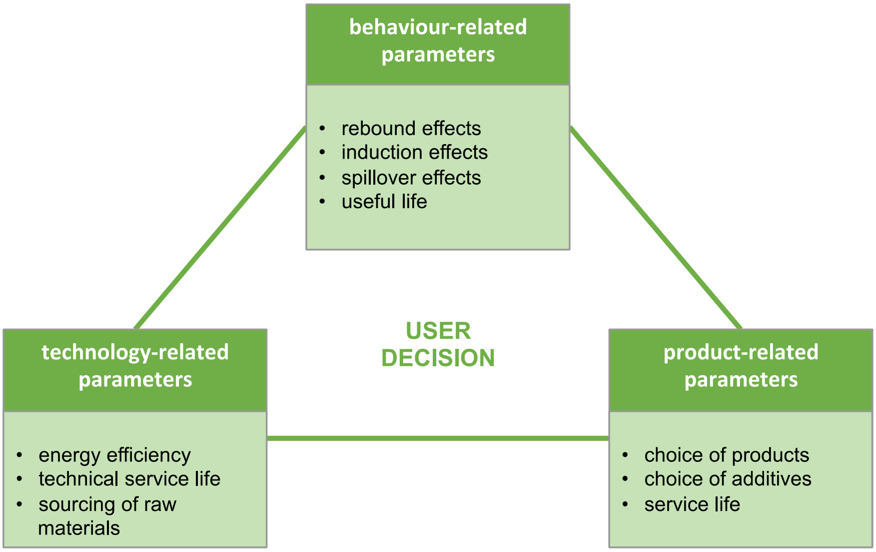

Fig. 1 User decisions affecting the environmental performance of products. Own work adapted from [4, 12]

ronmental effects in a life cycle assessment (LCA) study, user decisions and user behaviour should also be included in the modelling of the product system and the use phase. This is particularly the case if a significant part of the environmental impact results from the use of the product, for example through energy demand in housing and mobility [5, 6]. Further requirements for the definition of product systems and use phase result from the environmental assessment of services, product-service systems (PSS) and sharing practices, due to a stronger focus on use. Additionally, the quantitative environmental assessment of households and lifestyles demands a strong focus on activities and social practices [7–9].

Generally, user decisions have an effect on product-related parameters (choice of product, choice of additives and electricity grid mix) and on technology-related parameters (efficiency of the machine, other technical properties) [4]. Furthermore, secondary effects of technological change, such as rebound effects, can also lead to changes in user behaviour and thus in the environmental performance of the corresponding technology [10–12]. As a third point, behaviour-related parameters can be identified (see Fig. 1). With regard to sharing practices, both the intensification of use (e.g. carsharing; [11]) and the extension of useful life (e.g. clothing exchange; [13]) can become a decisive factor for the environmental performance. In addition, further spillover effects resulting from a fundamentally critical examination of consumers' consumption behaviour can become ecologically relevant for assessing eco-innovations at the household level.

In order to advise decision-makers in politics, businesses and the consumers conscientiously and to identify opportunities to improve environmental performances, user behaviour has to be a prominent part of environmental assessments. However, practice shows that LCA studies often either assume average usage parameters [14], or only address a limited number of life cycle phases ('cradle to gate'), without considering the use phase. The rather low priority given to the use phase in LCA is therefore also referred to as one of the 'key gaps' in LCA studies [4, 6, 15]. The causes are manifold. One of the reasons is certainly to be seen in the traditional focus of LCA on the product (and production) itself [16, 17]. Additionally, insufficient access to realistic (usage) data can be decisive [5]. Therefore, we aim to shed light on the relevance of user decision and behaviour in LCA and to emphasize relevant modelling aspects.

The remainder of the chapter is structured as followed: In Sect. 2 we discuss aspects of modelling alternative use and collaborative consumption with regard to the definition of goal and scope of LCA studies. We then concentrate on the use phase with a focus on rebound effects and the collection of inventory data for user behaviour in Sect. 3. Subsequently, we critically reflect processes of decision-making in the context of LCA in Sect. 4. The chapter ends by outlining further research needs in the concluding Sect. 5.

2 Modelling Alternative Use and Collaborative Consumption Patterns

As an effect of dwindling resources and increasing greenhouse gas (GHG) emissions a transformation of conventional production and consumption patterns is necessary to decrease the impacts on the environment. In recent years, new models of consumption and production have been developed that can be assigned to the *Sharing Economy* or *Collaborative Consumption* [18, 19]. As a particular consequence of the digital transformation, numerous business models have been developed that enable users to easily access products without necessitating ownership. The sharing economy is seen as a special opportunity to make products more environmentally friendly [20]. Another important strand of literature dealing with alternative consumption patterns are Product-Service Systems (PSS). PSS are defined as "a marketable set of products and services capable of jointly fulfilling a user's need" [21] and are considered to have potential environmental benefits [22]. There is still a small number of LCA studies dealing with PSS [23] but the methodological challenges are well documented and some suggestions to help overcome them have been made [24, 25].

Due to the fact that within collaborative consumption models, single products are used by multiple users, it is postulated that products are used in a more efficient manner and therefore less product units are necessary to meet the demand. As fewer products are produced, resources can be saved and hence emissions reduced. This raises some LCA modelling questions which we will discuss bellow. On the one hand, the method with which business models of collaborative consumption can be represented in LCA studies and, on the other hand, which product parameters change through alternative consumption patterns.

2.1 Defining Use and System Boundaries of Collaborative Consumption Models

By applying LCA, the question of environmental sustainability of new business models and collaborative consumption patterns shall be addressed. To assess the environmental impacts of new business models with LCA, the challenge is more complex as changes in demand have also to be addressed.

Those issues already begin to appear in the first phase of an LCA. Here, the object of investigation must be defined. If we understand the sharing economy as a business model, one approach is to define the business model as a "technical system". A business model describes the value-creation logic of an organisation and includes the organisation, the processes to create its outputs and the outputs themselves [26].

Considering the offered services and ways to use products within the sharing economy or within the field of collaborative consumption as a business model, the question remains as to how said model can be adequately translated into an object of investigation for an LCA. For this purpose, we identify three different perspectives:

1. **Product or service perspective**: The **first perspective** is the most common LCA perspective. In this case the sharing economy business model and its specific application has to be modelled analogously to a product system. The character of the assessment is highly product-related which can lead to neglect of the use phase.
2. **Organisational perspective**: The **second perspective** tries to assess the sharing organisation itself and therefore aims to calculate the organisational environmental impacts of the provider.
3. **User or household perspective**: The **third perspective** takes user behaviour into account. If more than one person is analysed, the system can be described as the consumption behaviour of a household.

Following this argumentation, we see that the sustainability potential of the sharing economy is largely determined by user behaviour. The question arises whether the promised environmental potentials are met or not, for example due to a boost in consumption activities or due to the use of more efficient products. Only the third perspective ('user or household perspective') also allows behavioural changes to be considered and thus enables rebound effects to be taken into account. We will discuss the nature of possibly occurring rebound effects later in Sect. 3.

However, if we argue that the use phase is crucial for the environmental potential of a new business model, then the use must be adequately defined in the first stage. Technically spoken, the use is the output of a product system. In LCA the use is defined by the determination of the functional unit. The functional unit normally is the key function of a product, expressed in a quantitative and measurable number. The functional unit of 'car' can be described as a certain number of kilometres a person is transported during one year. Consequently, the function of a car is the transport of a person from A to B. The definition of the functional unit is one of the key steps of an LCA (see Sect. 4). One of the most common errors in LCA studies is that comparisons are made which are not based on the same functional unit [27]. This is the reason why the functional unit is primarily chosen with the purpose of comparability at the expense of accuracy. Differentiated consumption patterns as they occur in the field of the sharing economy or collaborative consumption are more complex and need a more specific definition of their functions and benefits.

Furthermore, those new models usually offer more than only one function. In the case of car sharing the function of transporting a person from A to B must for example be supplemented by the temporal and geographical availability of a suitable vehicle. Beyond further technical functions a variety of social functions have to be taken into account as well. Those aspects are also described in Sect. 4. In the context of assessing new business models in the field of collaborative consumption and sharing economy, the integration of multifunctionality is crucial to enable a holistic approach.

2.2 Implications of Alternative Consumption Patterns for the System Under Consideration

Alternative consumption patterns such as the use of second-hand products or the shared use of goods have impacts on certain product parameters as well as systemic effects which must be considered when conducting an LCA study. Second hand products like clothes will have a prolonged service life as they are used by various consumers consecutively. On the contrary, a car used in car sharing will most likely be used for a few years less than a car only used by a private household. If several people share a car, the drive mileage of that car per year will go up and it will need more maintenance. In addition to the influence of use intensification on the car, the emissions of greenhouse gases and pollutants will increase if users use a car more often instead of using alternative forms of transport like e.g. public transport or bikes.

Zamani et al. [13] assumed a doubled and even a quadrupled number of uses for clothes offered in fashion libraries compared to their baseline scenario. In addition to that, they modelled different customer transportation scenarios. Their results show a huge influence of the mode and distance of transport which would be neglected if they would only assess directly product related impacts. Firnkorn and Müller [28], while not conducting a full LCA, took a broader view on the implementation of a free-floating car sharing system in a German city. They used primary survey data to forecast the effects on private vehicle ownership in cities. Both examples show that it is of great importance to include processes and effects which are, at first glance, outside the business model.

Obviously, this has an influence on the way researchers and practitioners must assess products used in PSS or other collaborative consumption schemes. They should collect additional data (or must make assumptions based on literature, see Sect. 3) and define goals and scope in a way which allows comparison with the alternative consumption patterns with other business models like conventional consumption (purchase, use and disposal of goods).

3 Inventory Analysis and the User Behaviour

Modelling user behaviour describes how the product system is actually used with regard to its function and includes, in particular, findings on the duration and intensity of use of the product system, on the consumption of energy, water and other auxiliary materials during the use phase, and on waste production [4, 6]. Additionally, we must recognise that products are used by actual users and that those users are affected by their choice and use of a product beyond the analysed product itself [12]. By describing the use phase in a theoretical way, we can distinguish three components of a use phase model: (1) product-related parameters required during the use phase; (2) intensity of use of certain product parameters during a certain action; and (3) frequency of user actions [29]. All these modelling parameters combine to provide information regarding intensity and magnitude of certain unit processes during the use phase. Thus, data on user behaviour and the use phase are not part of life cycle inventory, but are "inventory-related data" [27]. In the following, we will focus on two aspects that have been largely neglected so far in modelling of the use phase. First, we will discuss the role of rebound effects when assessing user behaviour. We then analyse, which types of data sources are needed for a more realistic modelling of user behaviour in LCA.

3.1 User Behaviour and Rebound Effects

We have already discussed that the user affects the environmental performance of a product or service. However, it can also be observed that the use of a product affects the user in his/her behaviour as well, which again is potentially of environmental importance. An example: the way in which a car is used, e.g. car sharing or avoid speeding, affects the environmental implications of driving. But using car sharing and avoid speeding also affects the user's expenditures, time budget and mindset, which leads to changed consumption patterns (e.g. the car sharing user may use the saved money to fly to a remote island). Such rebound effects can substantially decrease the environmental potential of innovations to a point where the rebound effect is even bigger than the direct savings. Despite specific factors that are often associated with rebound effects, e.g. time and money, they can be more generally described as unintended consequences due to changes to reduce environmental impacts [30]. While mentioned factors might seem to describe a rational behaviour of allocating newly gained resources (time or money) to new (indirect rebound effects) or intensified (direct rebound effects) activities, rebound effects must also be described with sociological and psychological effects, such as *bounded rationality* [31] and other theories of decision making.[1] Santarius and Soland [43] describe the changes in con-

[1] Ajzen [32] introduced the Theory of Planned Behaviour, which conceptualizes decisions as actions maximising favourable outcomes for the self. However, in recent years alternative influences of decision making become increasingly prominent. Apart from rational evaluations of different options,

sumer preferences for the use of a certain good as motivational rebound effects and explain those motivational rebound effects with the concepts of *diffusion of responsibilities*, *moral licensing* and *attenuated consequences*. On a macro level, rebound effects have been well known since at least 1865, when Jevons described in his book "the coal question" the increased demand of coal in the UK due to the work of James Watt on the steam engine, which increased its efficiency threefold (Jevons' Paradox). The economist Staffan Linder [44] analysed the increased scarcity of time due to increased productivity. The corresponding economic wealth that increases opportunities for consumers while only gaining a small amount of leisure time, where actual consumption can occur, leads to a "harried leisure class". So again, gains in productivity are not used to decrease the demand in resources, but to intensify economic throughput. This acceleration of the economy and life itself due to rebound effects [45, 46] is at the centre of the question on how can we decrease global environmental impacts instead of just shifting problems.

While rebound effects are not new to the scientific community, they seem to be widely neglected in the field of LCA [47, 48]. For example, the ISO standard gives no advice on how to address rebound effects [49]. The most common argument is the increased complexity, as products and activities have to be considered that are not part of the actual value chain of the product, service or PSS under consideration. However, the International Reference Life Cycle Data System (ILCD) Handbook identifies rebound effects as part of secondary consequences that should be included in LCA and proposes a consequential modelling approach [27]. A few modelling approaches can also be identified in the literature that integrate different types of rebound effects into LCA [11, 17, 50–52]. However, despite methodological challenges that are being addressed, finding and gathering suitable data on user behaviour and rebound effects remains a problem for LCA practitioners.

3.2 Data Sources and Methods of Data Collection for Modelling of User Behaviour

The required overall data quality of the LCA study is crucial for the identification of data and information needs and adequate data sources. Two general data sources in

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we increasingly identify a discussion of value-based theories that consider the individual's duty to act in a certain way. One of such examples is the Value Belief Norm-Theory proposed by Stern [33] that mostly focuses sustainable behaviour patterns and draws a decision paths moving from values, over different beliefs (e.g. worldviews) to the individual's personal norm that all together guide her actions. In addition, the role of emotions in human decision making is increasingly noticed within this field of decision research [e.g. 34–36] and in particular within sustainable science [37]. Recently we also see a development of theories that consider behaviour as automatic processes, learnt reactions, habits or unconscious associations, which all impact the individual decision process [38–40]. However, decision can also be influenced by external factors such as law and context conditions [41], implying a choice architecture [42], which considers informal and physical environments as prerequisite for decision making. All of these approaches highlight the complexity of human decision making and the challenges to involve these processes in LCA studies.

LCA studies can be distinguished: (1) primary data for specific cases, such as measurements, and (2) existing (primary and secondary) data sources, such as statistical data, LCI databases or product-related data from the producer/supplier. The criteria for data quality are derived directly or indirectly from the objective of the LCA study and include accuracy, completeness, and uncertainty/precision [27].

Typically, LCA studies rely on aggregated secondary data and average usage parameters to model usage behaviour [14]. By using statistical data, potentially different user behaviour is neglected. As a consequence, the LCA results are associated with a high degree of uncertainty concerning the actual user behaviour and the resulting environmental impacts. This is particularly relevant in cases where the use phase is responsible for a large part of the total environmental impact of the product system [5, 53].

This is why several LCA studies have already taken individual behavioural variations into account, e.g. by modelling different usage scenarios, by using sensitivity analysis or by using regionally disaggregated data [1, 4, 6]. In a study comparing the environmental effects of three types of nappies (disposable, home-washed reusable and commercially-washed reusable) the authors used scenarios to model the variability in user behaviour for usage rates and washing practices [54]. In this way, it was possible to identify certain environmental determinants in behaviour. Furthermore, it was possible to derive areas of action in which a reduction of the environmental impact can be achieved. However, the modelling of different usage scenarios was based on assumptions and on best practice [54]. Shahmohammadi et al. [4] used regionally disaggregated data to analyse the variability in GHG emissions from washing laundry in 23 European countries. In addition to regional and product related environmental data, the authors also used data from a European consumer survey on product usage and washing habits. The variability in user behaviour manifested in country-specific differences in the type of detergent and washing temperature. Depending on the electricity mix, the authors found different consumer choices to be the dominant source of variability in the results [4]. In a study comparing the print and tablet editions of a Swedish magazine, sensitivity analysis was used to identify the impact of the changed user behaviour of the tablet users on the overall environmental results [1]. However, the modelling of reader practice was partly based on secondary data and assumptions [1]. Applying data quality criteria to the modelling of user behaviour, uncertainties regarding the use of a product can be described as *parameter uncertainty* (lack of data) or *variability in objects/sources* [5, 55]. In the studies described above, it becomes clear that assumptions are used in cases where there is no precise information on user behaviour. To ensure data quality, these assumptions should at least be verified by secondary data or previous research [5].

For a more realistic modelling of the use phase, user choice and behaviour, Polizzi di Sorrentino et al. [6] call for a stronger inclusion of behavioural science in LCA. Particularly the use of surveys for self-reported behaviour and of sensor technology for direct behaviour measurement could be useful for scenario modelling, inventory data collection or impact assessment. In addition, Daae and Boks [5] propose further supplementary methods for a realistic modelling of the use phase such as expert interviews or simulations. Moreover, transdisciplinary research approaches, in real-world laboratories or sustainable living labs for example, provide an opportunity to collect specific data about user behaviour [56, 57]. Table 1 summarises suitable existing data sources and methods of data collection for a more realistic modelling of user behaviour in LCA. However, even though these approaches have been tested generally, their combination with LCA is sometimes of a more theoretical nature.

4 Processes of Decision-Making in the Context of LCA

In the following section we argue that not only potentially different user behaviour is a determinant for LCA uncertainties, but also the decision process within LCA itself might be a source for the identified limitations. Although the procedure of an LCA study is defined in the ISO standards [49], the four phases of an LCA also involve decision-making processes which give space for personal interpretation.

First, the definition of the goal and scope of the study includes many decisions, e.g. regarding the functional unit, the system boundaries and limitations. Although methodological and technical details are defined in the ISO standard, the decisions concerning the individual study remain eventually subjective. Also, the inventory analysis obtains subjective aspects. By defining inventory flows and allocation rules, the decision-maker might need to simplify the "real world". Therefore, it might be worth considering a selection of parameters, for example based on data access, personal preferences concerning the understandability, or the importance ascribed to the items [6]. This focus on specific items in turn might influence the overall LCA. The impact assessment also obtains personal decisions of the person conducting the LCA. In addition to the selection of impact categories and the underlying calculation method, these might also include vaguely defined elements such as normalization, grouping and weighting. While most LCA practitioners clearly try to accomplish as much transparency as possible within their studies, these steps are often based on personal experience, knowledge and other contexts. Also, the interpretation of LCA results is not only based on the former phases but also includes subjectivities. It is one of the toughest steps of an LCA process, as it involves an understanding of every single parameter in regard to the goal and scope of the study and the identification of most significant elements. Still, it is a personal evaluation that again is affected by the background of the decision-maker. Following this description of an LCA process we clearly identify interspecific and intraspecific interdependencies, as every phase influences the other and different steps within one phase are related to one another [61]. This critique is also reflected by studies that find different results for the same products calculated with LCA when conducted by different persons and with different methods [62, 63].

To counteract such critique—and to further emphasize LCA as a tool to support environmental-friendly decisions—we need to think out of the LCA-tool-box. We not only have to unlock traditional product-focused LCA thinking by more strongly integrating user behaviour (see Sect. 2), but also overcome the "power" of the decision-maker. Here, participatory and transdisciplinary approaches seem a promising tool to

Table 1 Primary and secondary data sources for modelling user behaviour in LCA

Approach	Details and references
Primary data—data collection	
Sustainable living lab	Sustainable living labs (SLL) provide a research infrastructure and methodology to test potentially sustainable innovations in real-life scenarios [56]. The user- and stakeholder- integrated approach allows the collection of technically relevant insight data and on user behaviour, especially for PSS. An SLL consists of three steps, the *insight research*: household analysis before the intervention, the *prototyping*: development of scenarios and prototypes and the *field testing* of such prototypes. The analysis of actual consumption patterns also allows the analysis of the unexpected utilisation of products—rebound effects [58]
Real-world laboratory	Real-world laboratories (RWL) are social arrangements consisting of the interaction between people, their environment and their social and institutional interdependencies [59]. The complexity of such a setting allows a comprehensive analysis of interdependencies and impacts in the real world. Since RWL are of inter- and transdisciplinary nature, LCA practitioners do not just do their own RWL in order to receive data on consumption patterns, but can participate in a RWL that consists of a broader team. In comparison to SLL, RWL do not necessarily have a single product or a specific innovation at its centre but rather a part of the society as such. RWL are often used in the field of urban transition, even though other research areas are not excluded [57]
Household analysis	Household analyses provide a broad picture of consumer behaviour and lifestyles on the micro-level. As in SLL, the look beyond a specific product or PSS allows identification and consideration of rebound effects. In household analysis, surveys concerning all household components (e.g. nutrition, housing, goods, mobility, leisure, other) are being conducted [8]. In this way, reduction potentials of an innovation or behaviour change cannot only be assessed on a relative scale, but the relevance for absolute reduction targets can be identified, too. A household analysis can be accompanied by e.g. the assessment of socioeconomic data and road mapping processes to decrease environmental impacts in the future [7]. Online surveys of households can increase the amount of collected data drastically [60]
Behavioural science support	Behavioural science can for example support the modelling of different user scenarios in LCA based on typologies of use, lifestyle, geographical context, demographic aspects or income. Behavioural sciences could also be useful for inventory data collection. In addition, they could also be used to identify various behaviour-related exposure scenarios in the use phase for the impact assessment. Particularly surveys for self-reported behaviour and sensor technology for direct behaviour measurement appear to be suitable for integration into LCA [6].

(continued)

Table 1 (continued)

Approach	Details and references
Measurements	For data on the use stage of consumer products, the ILCD handbook proposes data measurements at the operated processes [27]. Measurable parameters could be, for example, energy consumption, emissions or useful life
Other supplementary methods	Additional supplementary methods such as expert interviews, peer reviews or correlation and regression analysis can also help to address variations in user behaviour [5]
Secondary data—existing data sources	
Household panels	National or international household panels that repeatedly gather e.g. socio-economic, geographic and demographic data. The German SOEP (socio-economic panel) interviews nearly 30,000 people in 11,000 households every year
Income and expenditure statistics	Income and expenditure surveys gather data on the spending of households. Additionally, socio-economic data are gathered which allow peer-group specific analyses. In this way income elasticities for each income level can be identified and used to calculate economic rebound effects [9, 11]
Previous interdisciplinary research	Findings on specific user behaviour from previous interdisciplinary research can be used as a reference to justify assumptions in user behaviour modelling [5]

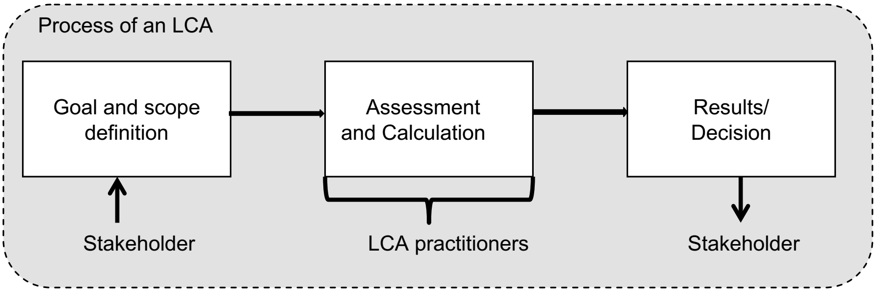

Fig. 2 Integration of stakeholders into the process of LCA modelling

overcome personal bias and generate a more realistic modelling of user decisions and behaviour. However, as described in Sect. 3, recent LCA approaches integrate—if any—behavioural aspects based on empirical data or own (personal) assumptions even though it has been proven that the best way to predict behavioural intentions is by asking people directly [6] as promised by participatory approaches. Additionally, it has been proven that consumption obtains other functions than merely the market activities of consumers. It further supports individuals in expressing personal attitudes, interests, identities and social roles to others, also known as signalling or symbolic consumption [64]. Those dynamics might have an impact on the frequency and duration of use as well as on the function of a product. For example, Haucke [65] found that the Fairphone, a sustainable smartphone, was not only used based on its functionality but more importantly it was seen as a symbol to express membership in sustainable oriented groups and a sustainable attitude in general. This further included various behaviours such as community activity in order to support repairability practices, bartering of components and knowledge concerning energy efficiency and other technological improvements.

Alternative approaches to data collection such as sustainable living labs and real world-laboratories (see Table 1) might be a fruitful path to support LCA practicioners in their decision-making process. However, it is worth mentioning that in many cases, such alternatives are hardly practical, due to limitations of time, resources and access. Supplemental to user integration, the focus on the product itself might lead to an oversight of various influencing parameters since attitude dynamics and user behaviour on the consumer side are only one aspect of the use phase. Although technological, political, legal, economic and scientific developments possibly influence the LCA, they remain another black box. Integration of actors with expertise in these areas could help to overcome such shortcomings. Figure 2 illustrates a possible integration of stakeholders.

While stakeholders might not contribute to the inventory analysis itself, they notably could contribute to the definition of the goal and scope of the study. In this regard the LCA could assure a holistic understanding of the product under study. However, it might be the interpretation phase where "real-world-thinking" is most significant. When it comes to the evaluation of the performance of a product or service, it is of utmost importance to have knowledge of the areas affected. For example, while the automobile was hailed as a public health innovation and a major improvement to urban living as well as environments during the time of its introduction (compared to horse manure and rotting dead horses in the metropolitan streets) its consequences such as condemnations of gas-guzzling, greenhouse-gas-spewing, status-symbolizing SUV's and so on, had not been considered if they were known at all [66]. While at that time, environmental knowledge was at its beginning, scientific development today makes it easier to discuss possible environmental and social impacts of products. However, following our argument, we see a need and a huge potential to discuss results of an LCA among disciplines and with stakeholders in order to upgrade the LCA method towards a holistic approach.

5 Conclusion

In this chapter, we aimed at examining the relevance of user decision and behaviour in LCA and to highlight relevant modelling aspects. With regard to the environmental assessment of alternative use and collaborative consumption, modelling aspects for the definition of system boundaries, the definition of use and further implications on service life and useful life were discussed. Furthermore, the role of rebound effects when modelling the use phase were addressed and primary and secondary data sources for a more realistic modelling of user behaviour in LCA were presented. In addition, processes of decision making in the context of LCA were reflected upon and the relevance of stakeholder integration in the modelling process was highlighted.

When arguing that LCA is a tool to make decisions to increase global sustainability, variances in user behaviour are not just a factor that should or could be considered when analysing the environmental effects of a product or service. LCA practitioners have to consider changing product properties as well as impacts on consumer behaviour which seems to be outside the traditional scope of a study. In order to do so, interdisciplinary approaches of collecting primary data should be considered. Especially in the context of new forms of consumption, where fuzzy system boundaries and product functions are well known, we identify a need for multi-perspective integration to clarify such a complexity. Further research is needed to overcome such limitations and to design new methodological approaches of transdisciplinary and participatory integration of stakeholders within LCA studies.

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Ecological Assessment Based on Environmental Product Declarations

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Abstract For some years now, environmental product declarations (EPDs) have been available to the public mainly for b2b communication. These contain important environmental impact categories (EIC) and are based on an externally audited life cycle analysis (LCA) according to current ISO standards. The information from EPDs can be used for company-internal product benchmarks and ecological optimizations. The method described in this paper follows the German Federal Environment Agency's ecological priority method and investigates ecological scarcity, distance-to-target and specific contribution. The article explains the ecological single-score method based on EPD data and validates it against common single score assessments (ReCiPe and UBP). Therefore, 9 case studies have been examined by comparing the full LCA with the results of the new method based on EPD data. As a result, 8 out of 9 studies have similar results with the same benchmark rating. With this information important action alternatives for the future can be derived and ecological product optimization can be pursued.

Keywords Eco-efficiency analysis (EEA) · Environmental product declarations (EPD) · Single-score analysis · Environmental benchmarks · Ecological priority

1 Introduction

Ongoing environmental protection problems are increasingly bringing the importance of environmental issues to the foreground. The causes are spread across several environmental impact categories and pose problems to interested parties, if they do not possess an LCA background. Type III declarations, also known as EPDs, are currently a frequently used LCA format. There are now more than 5500 EPDs, published by various programme operators in accordance with DIN EN ISO 14025 [1, 2].

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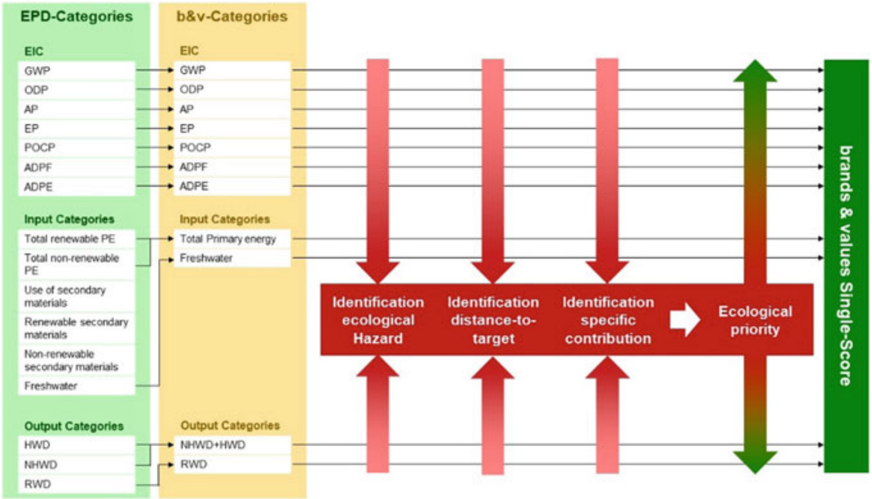

Fig. 1 Process scheme of the b&v method

To date, they are mainly used in the construction industry as a tool for communication and are also required for green building certifications e.g. to calculate the buildings LCA.

LCA results of EPDs include a variety of EIC (see Fig. 1) and Input/Output indicators, which are specified by the DIN EN 15804 [3]. According to DIN EN ISO 14040 [4] and 14044 [5], the standard for LCAs, an assessment requires a suitable set of EICs that depend on the investigated product system. DIN EN 15804 specifies this set for all products in the construction sector in advance, regardless of the nature of the product.[1]

Despite potentially limited information value compared to a full LCA according to ISO 14040 & 44, a new assessment method has been developed to make use of the great availability of EPDs. This single score evaluation system based on the EPD results follows the ecological priority method of the German Federal Environment Agency [6] assessment by considering political target values (distance-to-target), total emissions (specific contribution) and potential damage (ecological hazard).

The publication deals with the methodology as well as the examination of the results. Using nine case studies, our method will be validated against the standard single score assessments methods ReCiPe and UBP.

[1]An update of the DIN EN 15804 in 2018 extends the amount of EICs. The described method has not yet been extended for this update.

2 UBA Ecological Evaluation Method

In 1999, the UBA [6] published 'Bewertung in Ökobilanzen [...]'. The environmental impact categories examined are neither quantitatively nor qualitatively comparable and were hierarchized based on their influences on the endpoints 'human health', 'structure and function of ecosystems' and 'natural resources' on the basis of the normalization methodology developed. The more the following criteria influence the protected goods and the more the environmental impact applies to the following criteria, the higher the priority is given to the environmental impact category. The assessment leads to a relative ranking of the categories from 'A' (highest ranking) to 'E' (lowest ranking).

<u>Ecological hazard</u>

The ecological hazard assesses the hazard of the impact categories to each other (independent of current health status). Here are the following indications for classification. The higher the hierarchy,

- the more endpoints are affected,
- the more the individual endpoints are affected,
- the more irreversible the damage in the endpoints is and
- the further the damage can spread (globally or regionally).

<u>Distance-to-target</u>

The further the environmental status of an impact category currently deviates from an ecologically sustainable status, the higher the hierarchization. The UBA mentions the following indications for hierarchization:

- The distance between the current environmental status and a quantified quality objective. The distance between the two values is expressed as a quotient and can be compared. The higher the quotient, the higher the rating.
- If there is no quality objective for the environmental impact category, an environmental action objective can also be used. This is defined as the difference between the current load and the maximum permissible load. This results in a reduction quantity by which the emissions within the category would have to be reduced to meet the action target. The higher the reduction, the higher the rating.
- Current trends are also taken into account: If there is an increase in emissions instead of a reduction, the category is rated higher.
- If actions have been presented for the reduction of emissions, these have an impact on the hierarchies, depending on their feasibility. For example, it is very difficult to implement a societal lifestyle change, if this is the only way to reduce emissions.

<u>Specific contribution</u>

- Measures the (absolute) influence of the impact indicator result on the reference value (e.g. annual GWP emissions in Germany). The largest relative contribution receives the highest hierarchization, the other environmental impact categories are subordinate to this value.

LCA results are only required for the specific contribution criterion. The highest relative contribution receives the highest priority, all other categories subordinate to this value in 20% steps. If 2 or more LCAs are compared with each other, the respective (relative) smaller value of each environmental impact category applies for the determination of the specific contribution. The maximum value of the smallest contributions thus represents the 'A' rating.

The other criteria are hierarchized once and then apply across examples for a certain period of time (2 years are recommended, then the hierarchization should be generally revised and adapted to current environmental situations). The highest or lowest priority does not necessarily have to be assigned here.

The UBA method does not sum up the values but ranks the overall result once again into the priorities 'very low' to 'very high'. In this case, the UBA chooses a verbal-argumentative approach and does not generate a single score, such as ReCiPe [7] or UBP [8] output.

3 b&v Method

The UBA's [6] approach of ecological priority was applied in the development of the b&v method. The UBA evaluated other EICs at that time, but there were some overlaps (see Table 1). The approach was transferred to the EPD EICs and indicators (see Fig. 1).

In contrast to the UBA method, the result of the b&v method is available as a single score. For this purpose, the ecological priority (verbal-argumentative approach: from

Table 1 b&v and UBA assessment of distance-to-target and ecological hazards

		Distance-to-target 1–5		Ecological hazard 1–5	
		b&v	UBA	b&v	UBA
GWP	kg CO_2-eq	5	5	5	5
ODP	kg CFC11-eq	3	2	4	5
AP	kg SO_2-eq	4	4	3	4
EP	kg PO_4-eq	3	3–4*	3	4*
POCP	kg Ethen-eq	2	4	2	2
ADPE	kg Sb-eq	2	–	2	–
ADPF	MJ	4	2**	3	3**
∑primary energy	MJ	4	–	3	–
Freshwater	m^3	1	–	2	–
HWD + NHWD	kg	2	–	2	–
RWD	kg	2	–	4	–

* EP at UBA is divided into aquatic (distance-to-target 3) and terrestrial (distance-to-target 4)
** is described by the UBA as EIC "Knappheit fossiler Energieträger"

Table 2 Initial table for the calculation of the specific contribution

EIC (Environmental impact category)	Product x	Product x + 1	(National) threshold
GWP	Y_{GWPx}	Y_{GWPx+1}	$Y_{MAX(GWP)}$
ODP	Y_{ODPx}	Y_{ODPx+1}	$Y_{MAX(ODP)}$
AP	Y_{APx}	Y_{APx+1}	$Y_{MAX(AP)}$
EP	Y_{EPx}	Y_{EPx+1}	$Y_{MAX(EP)}$
POCP	Y_{POCPx}	$Y_{POCPx+1}$	$Y_{MAX(POCP)}$
ADPE	Y_{ADPEx}	$Y_{ADPEx+1}$	$Y_{MAX(ADPE)}$
ADPF	Y_{ADPFx}	$Y_{ADPFx+1}$	$Y_{MAX(ADPF)}$
Primary energy	Y_{PEx}	Y_{PEx+1}	$Y_{MAX(PE)}$
Freshwater	Y_{SWx}	Y_{SWx+1}	$Y_{MAX(SW)}$
NHWD + HWD	$Y_{N\text{-}HWDx}$	$Y_{N\text{-}HWDx+1}$	$Y_{MAX(N\text{-}HWD)}$
RWD	Y_{RWDx}	Y_{RWDx+1}	$Y_{MAX(RWD)}$

$Y_{MAX}()$: Limit values of a substance (Germany)
$Y_{x}()$: EIC value from the EPD or LCA of a product (sum of all declared modules from A1 to C4)
$Y_{s,x(max)}$: Highest specific contribution for each product
$Y_{s()}$: Smallest EIC-specific contribution of all products
$Y_{max,x()}$: Largest contribution of all smallest contributions
$Y_{prio,x()}$: Ranking for the hierarchization of each substance
$f_{x()}$: Scaling factor
$F_{rel()}$: Relative contribution to total factor
$Y_{N,x()}$: Standardized load for each EIC of each product
$Y_{FN,x()}$: Standardized weighted load for each EIC of each product
Y_{x}: Sum of standardized weighted loads; single-score value for the b&v method

low priority to very high priority) is transferred into a weighting scheme and provided with weighting factors.

Figure 1 shows the EIC and input and output categories that are mandatory according to the DIN EN 15804 in the green column and the adopted EIC and material flows in the yellow column.

To combine the results into a single score, the letters assigned by the UBA are converted into numbers (1 = E, 5 = A) and added (maximum achievable value is 15 = very high, lowest value is 3 = very low).

The table above shows the assessment of distance-to-target and ecological hazards of b&v and UBA in 1999. Due to limited space, the derivation for the b&v classifications are not considered in this publication. For decision-making purposes, particular attention was paid to the recommendations of the German sustainability strategy [9] and other national and international sources like German agenda for sustainable development. However, the strategy's compliance with the categories considered in the environmental assessment is only partial and can only support personal decision-making (Table 2).

First, the specific contribution of the products for each EIC is calculated in relation to the national limit values (1):

$$\frac{Y_{()x}}{Y_{MAX()}} = Y_{s,x()} \tag{1}$$

The specific contribution provides initial assistance for companies which should focus on EICs in order not to exceed any national targets in the future. Further calculation steps are needed to prioritize the specific contribution according to the UBA method so that the EICs can be ranked among themselves.

When comparing two products, the following procedure is used (2).

The smaller specific contribution for the scaling is selected for each EIC. If there is only one product examined, this is not necessary.

$$Y_{s,\min()} = \min\left(Y_{s,x()};\ Y_{s,x+1()}\right) \tag{2}$$

From the resulting list of smallest specific contributions ($Y_{s,min(GWP)}$, $Y_{s,min(ODP)}, \ldots, Y_{s,min(RWD)}$), the highest value is selected (Y_{prio}) and automatically receives the highest value to be assigned for the specific contribution (3).

$$Y_{prio} = \max\left(Y_{s,min(GWP)}, Y_{s,min(ODP)}, \ldots, Y_{s,min(RWD)}\right) \tag{3}$$

Based on Y_{prio}, a prioritization of all $Y_{s,min}$ values is now determined. This is done in the same way as for ecological hazards and distance-to-target on a scale from 1 (lowest value) to 5 (highest value). The steps to determine the scale are in 20% increments, with Y_{prio} automatically receiving the 100% value 5.

The following table shows an exemplary result of a product based on the EICs GWP, EP, freshwater and RWD. The specific contribution of the product is highest for the freshwater consumed. However, due to the high ecological hazard and distance-to-target values, GWP is the category with the highest ecological priority.

The specific contribution determined as well as the values for distance-to-target and ecological hazard, can now be combined to form a single score, the ecological priority. The value range can then be given an additional weighting factor. For the b&v method, the weighting factor e^x is used.

This weighting factor expresses the meaning or the "distance" between the respective hierarchy levels between A and E.

The Euler number (e^x) forms the basis, for x as exponent values from 1 (E = very low ecological priority) to 5 (A = very high ecological priority) are used corresponding to the ecological priority classification. Compared to a linear progression (e.g. 1, 2, 3, 4, 5) this underlines once again the ecological urgency of an environmental impact category in which all categories of ecological priority have been rated high or very high.

Table 3 Exemplary composition of the ecological priority

	Specific contribution $Y_{prio()}$	Ecological hazard	Distance-to-target	Ecological priority b&v
GWP	2	5	5	12
EP	2	3	3	8
Freshwater	5	2	1	8
RWD	1	5	2	8

Table 4 Selection of the scaling factor

Value range (ecological priority)[a]	Interpretation of the UBA method	$f_{x()}$				
		X	2x	2^x	e^x	User defined
1–3	Very low	1	2	2	2.72	
4–6	Low	2	4	4	7.39	
7–9	Medium	3	6	8	20.09	
10–12	High	4	8	16	54.60	
13–15	Very high	5	10	32	148.41	

[a]The value range differs slightly from the UBA classification. b&v classifies linearly (3-steps)

Table 5 Usage of the scaling factor and calculating the relative contribution

	Ecological priority b&v	$f_{x()}$	$F_{rel,x}$ in %
GWP	12	54.60	47.54
EP	8	20.09	17.49
Freshwater	8	20.09	17.49
RWD	8	20.09	17.49
Sum		$F = 114.87$	100.00

The ecological priorities of the example above (Table 3) are now getting evaluated with e^x as shown in Table 4, summed up (4) and then displayed as a relative number (according to 100%) in the right column of Table 5.

$$F = \sum f_{(GWP)} + f_{(ODP)} + \cdots + f_{(RWD)} \tag{4}$$

To determine a single score, the normalized environmental impacts ($Y_{N,x()}$) are additionally required for each product (5). They result as the quotient of the actual environmental impacts of each product and the respective maximum environmental impacts ($Y_{max,x()}$).

$$Y_{N()x} = \frac{Y_{()x}}{Y_{max,x()}} \tag{5}$$

Using $F_{rel,x()}$ and the normalized load $Y_{N()x}$, the normalized and weighted load is calculated for each EIC of each product (6). The sum of the standardized and weighted EICs of a product results in the single-score result (7), which represents the environmental assessment in the b&v method:

$$Y_{FN,x()} = \frac{Y_{N,x()}}{F_{rel,x()}} \tag{6}$$

$$Y_x = \sum Y_{FN(GWP)} + Y_{FN(ODP)} + \cdots + Y_{FN(RWD)} \tag{7}$$

The evaluation does not necessarily require a second product for a benchmark (see Fig. 5). During the evaluation, the user receives information about the environmental performance, which shows them hotspots of the evaluated product system.

4 Results

The validation of the method is based on various practical examples from different industries, for which a complete LCA is available in addition to the EPD data and which can therefore also be evaluated according to ReCiPe and UBP. Out of the 9 considered validation examples, the b&v single-score was in accordance with the other two evaluation methods in 8 cases. In the other project, the results of the UBP were not conclusive with each other, so that the b&v method could not agree with both methods (Fig. 4).

Figures 2 and 3 show examples from the metal and plastic industry respectively. Every symbol in the 3 figures below is one product. It might be comparison between different product variants, production alternatives for product optimization or a comparison of similar products of different manufacturers. While the results in Fig. 2 are approximately stable, the results in the plastic example are between stable and not stable. This applies to the b&v method, but also to ReCiPe and UBP. In Fig. 4, the results for b&v and ReCiPe are similar, while for UBP the red dot (value 0.68; second best result) is not displayed in the same order as in the other two evaluation methods (fourth best result for b&v and ReCiPe).

The results show that the b&v method achieves the same results as the standard methods ReCiPe and UBP. The results naturally differ to a certain extent, since the focus is different depending on the methodology used. UBP has exclusively focused on Switzerland, while ReCiPe has a strong distance-to-target focus. It can be concluded that the b&v method is more consistent with the other two methods, the further the distances between the individual results lie. The smaller the distances become, the more cautious the interpretation should be. With the integration and updating of the UBA 1999 method, the procedure for determining the single score is up-to-date.

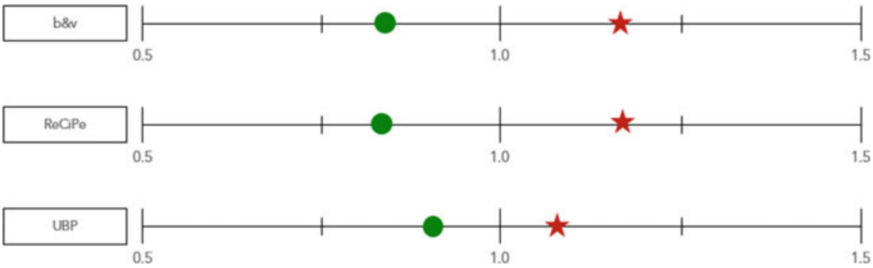

Fig. 2 Validation example metalworking

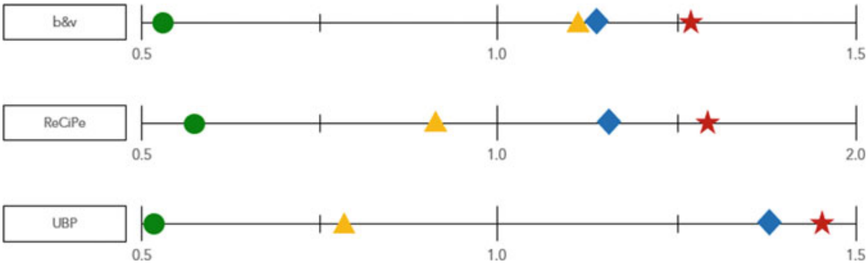

Fig. 3 Validation example plastics-production

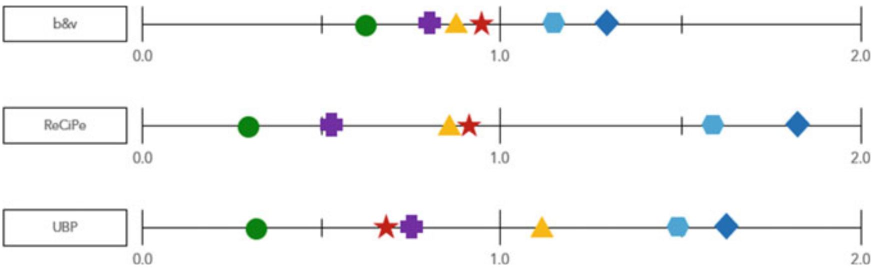

Fig. 4 Validation example wallcoverings

5 Summary and Outlook

According to study conducted by brands and values (publication in preparation), the satisfaction of EPD owners is high, but the cost-benefit ratio is not good because the content of EPDs is difficult to understand. They are mainly seen as an entrance ticket to the Green building market.

This may change with the b&v method, as important statements can now be easily deduced and illustrated and therefore be understood even by non-LCA experts. By using the method and raising the knowledge about environmental issues as well as providing company-intern benchmarks, the Type III declaration significantly increases the costumers benefit. In comparison with Type I [10] and Type II [11]

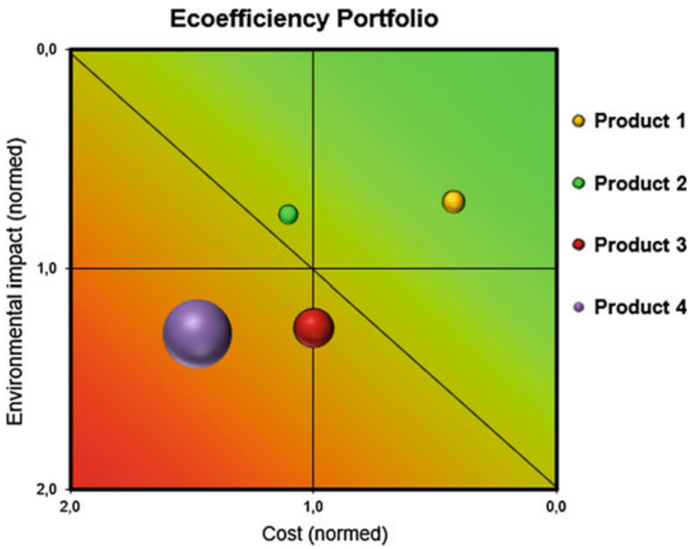

Fig. 5 Example of an ecoefficiency portfolio with 4 products

declarations, which can be used exclusively for marketing purposes, there is more added value possible.

The b&v single score can simply be included in an Ecoefficiency analysis (EEA). Therefore, it always needs at least 2 examples, which are getting compared. The single scores are displayed according to the BASF EEA scheme [12]. For this purpose, the mean value is calculated from the results and the relative distance of the points from the mean value is used (8 and 9). This ensures that the mean value in the graph is always 1. The interpretation is based on the relative distances to the value 1. The lower the value, the better the relative environmental performance of the product:

$$Determination\ 1st\ point = \frac{Y_1}{\frac{Y_1+Y_2}{2}} \tag{8}$$

$$Determination\ 2nd\ point = \frac{Y_2}{\frac{Y_1+Y_2}{2}} \tag{9}$$

If the ecological single score is supplemented by economic data, an EEA like the one once developed by BASF AG can be produced and evaluated.

In Fig. 5, product 1 has the best environmental and economic (cost) evaluation, compared with product 2, 3 and 4. The size of the bubbles were used to display the market share. Product 4 has the biggest market share, but is the worst of the 4 examined products in terms of ecoefficiency. As a result, the company should start focusing on producing more of product 1 and less of product 4.

The calculation methodology has the potential to be further developed on the basis of UBA specifications and used for product benchmarks based on EPD data.

Currently, each of the three categories of ecological priority is weighted equally. An additional weighting factor could, for example, increase the influence of the specific contribution, depending on which of the categories is subjectively considered to be the most important.

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Social Sustainability as a Target Figure in Life Cycle Assessment: Development of a Catalogue of Criteria for Measuring the Social Dimension

Claudia Hösel, Christina Hesse and Rico Pestinger

Abstract In contrast to the ecological sustainability dimension, the social dimension in science is less concretized and operationalized due to the inherent characteristics of social themes. The subject areas of social sustainability are sometimes difficult to distinguish from one another and thus very complex, mostly affecting both the individual level and society as a whole. Companies that wish to monitor compliance with social aspects, are therefore faced with the challenge of mapping and thus developing clear criteria for describing the social sustainability dimension. In this article, currently effective frameworks, process guidelines and management approaches are analyzed in terms of content. The derived results show five main categories of interest in social sustainability, whereby each can be sub-divided into sub-categories. Only 17% of all criteria could be identified as quantified. In addition, it is determined that the allocation of available data in a company or supply chain is feasible for several categories or subcategories. On the other hand, the close link with economic indicators allows for flexible analysis of social sustainability based on existing data applicable to the developed set of criteria.

Keywords Social sustainability · Social indicators · Social life cycle assessment · Social sustainability management

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1 Introduction

When the German Enquête-Commission "protection of mankind and environment" pushed the procedure of life cycle assessment, it had only ecological analyses and the reduction of ecological impacts in mind [1]. In the meantime, since 1994 the methodology was refined from life cycle thinking to isolated footprint measures and extent life cycle assessments. Today, social assessment-criteria are also of interest and so called sustainability-life cycle assessments or product-related social life cycle assessments are carried out [2]. Those are complemented with applications suggested by the commission, such as cost-benefit-, risk- and scenario-analyses. Furthermore, approaches like willingness-to-pay and stakeholder-analyses are applicable for the assessment of social sustainability [3]. The challenge posed by social impact assessment comprises dimensionality on the one hand and lack of quantification of social fields of action on the other hand. Dimensionality for process-oriented analyses is considered demanding, because social aspects mostly address issues on a societal level, such as peacekeeping, international and interpenetrative equity. Similar to the division of ecological strategies on weak or strong sustainability, social sustainability can be separated according to fundamental assumptions. For instance, Opielka [4] distinguishes between skeptical, narrow, internal and wide understanding of social sustainability. While a skeptical sense of sustainability leads exclusively to legal compliance, the other three levels of understanding focus more on the task at hand. In the narrow understanding of social sustainability the term is interpreted as redistribution and conflict-reduction accompanied by thoughts on ecological sustainability. This narrow description of fields of action focuses on social conflicts on a macro-economic level, such as unequal distribution of natural resources. The internal view of social sustainability reaches beyond present and urgent issues to focus on interpenetrative equity. World society is notified as population of interest and the protection and preservation of its heritage is applied as the corresponding task. Finally, the wide understanding of social sustainability questions societies focused on economic growth and discusses the conditions of post-growth society.

Altogether discourses on social sustainability take restructuring of society in the course of transformation or, with the words of Beck [5], metamorphosis into consideration, showing the rather broad and global view. For this reason the social dimension of sustainability is also sometimes referred to as socio-cultural or societal [6]. Companies assessing their impact or the impact of the production of their goods and services on social settings, on the other hand, will not be able to operationalize such broad schemes of social sustainability. Their impact, however, is significant, and consequently the research field on social impact assessment gains importance. Sufficiently detailed parameters for assessment need to be developed in order to adapt the understanding of social sustainability to the view of companies as active influencers.

2 Social Sustainability as a Target Value in Life Cycle Assessment

In order to integrate social dimensions of sustainability into life cycle assessments, adaptivity of corresponding criteria shall be facilitated within to the process-oriented approach. Conventionally product life cycles, corresponding to utility and consumption values (referred to as functional units) consist of phases similar to raw mineral extraction, production, use phase, recycling procedures and disposal. Companies are asked to at least consider upstream and downstream procedures, at best whole product life cycles, already in the course of product development, in order to be able to estimate potential ecological impacts. The systematic assessment of material flows, as common in ecological assessments and evaluations of product life cycles [7], shall now be transferred to product-related social-life cycle assessment, whereby humankind is integrated as producers, consumers and cost factors.

Amidst the societal and thus holistic view of social sustainability criteria the company's point of view has to clearly be stated in the light of such assessments. Hereby, system boundaries are specified as follows: gate-to-gate, assessments are only carried within the company grounds; cradle-to-gate, any upstream processes of value increase including raw material extraction are estimated; cradle-to-grave included the complete product life cycle from extraction of raw material to disposal [8]. Approaches that reach beyond the factory gates and consider especially raw materials production and provision of outsourced services, will grant more accurate results on the life cycle and can be integrated into the global view on the world society that is seen as social sustainability. However, bearing in mind cost-benefits-considerations and increased readiness to act, where social grievances are directly perceived, gate-to-gate assessments are yet of big interest. Within the chain of value enhancement, the workplace functions as a "natural junction" between lice cycle assessment and social sustainability, according to Hansjürgens [9]. Common criteria for the quantified display of social conditions are e.g. accident rates and cases of illness.

A suitable set of categories, that is easily implementable for companies striving to generate resilient information and develop practicable measure for improvement, is needed for the integration of social issues into product-related life cycle assessments. With this idea in mind, the UNEP and SETAC combine the structural procedure of the ISO 14040 [10] with elements from stakeholder analysis, to draft categories of stakeholders in social life cycle assessment [11]. Additionally, impact categories are phrased and assorted to the defined stakeholders. This point of view is adapted e.g. by Sala et al. [12], who nevertheless stated that life cycle assessments analogous to ecological estimations cannot be carried out for two reasons: changes occur faster with the behavior of the company, and impacts in terms of social performance can be considered both positive or negative. Further challenges arise, when service providers contribute outsourced procedures to the value chain, and cannot influence the whole chain on the one hand, and bias it on the other hand, so that precise target figures are needed in order to comply with set standards in the chain of value enhancement. An example for this would be the sector of logistics.

Research on measurability of social sustainability mostly focuses on selective fields of actions. For instance, Popovic et al. [13] focus their investigations on social sustainability in terms of human right and humane working conditions. Based on a literature review they extract 31 quantitative indicators for measuring the compliance to human rights and humane working conditions along whole supply chains. Yet, they suggest their framework to be extended and adapted.

Barbosa-Póvoa et al. [14] analyze methods of ensuring social sustainability on an operative level and conclude that sustainability as a whole is illustrated mostly with economic and ecological indicators, while social aspects remain widely unconsidered. Ahi and Searcy [15] confirm the little attention to social sustainability in assessments. Dubielzig [16] puts this fact to the characteristics immanent to social issues: they are bipolar because they both address individuals and society as a whole, and also social issues are immaterial and thus significantly less operationalized. The transformation of social aspects into qualitative and quantitative criteria on the other hand, is crucial to the derivation of target figures, performance assessments and monitoring activities throughout complete value chains. Kühnen and Hahn [17] approach indicators on the measurement of social impacts of companies from a system theory perspective. They established the lack of a systematic measurement of social sustainability performance.

In practice, companies are provided with various frameworks, procedure guidelines and management approaches, like e.g. SA8000 [18], as a means of support in their way to monitoring social sustainability in their value chains. Existing approaches to the management of sustainability issues are diverse and only manageable with difficulty [16]. Some approaches focus exclusively on the ecological dimension of sustainability, e.g. environmental management standard ISO 14001 [19]. Others address management of all three sustainability dimensions, e.g. sustainability guideline ISO 26000 [20]. In order to establish criteria as decision-making tools, as well as target figures in performance assessments of social parameters, those documents serve as a theoretical foundation for practical approaches to social sustainability. Hereby the focus on specifically quantified criteria does not seem sufficient, because of bipolarity and immateriality of social issues. For an accurate display of the social dimension of sustainability in practical approaches, the research shall not be limited to quantitative criteria, since those only describe a part-section of the research field, but qualitative criteria is also be involved. The research question is stated as follows:

To what extent is the social dimension of sustainability substantiated by measurable—qualitative and/or quantitative—criteria in presently effective frameworks, procedure guidelines and management approaches?

3 Methodical Approach

The methodology applied is divided into four processes and described below in individual Sections.

(1) ***Process* 1—*Identification***: The first process is aimed to identify the approaches that are relevant to the management of social issues. The identification of relevant approaches is based on online research in multiple databases such as Elsevier/Science Direct, Scopus Database, and Springer. The keywords used in the research were: "social sustainability", "social performance measurement", "social indicators", "sustainability indicators", "sustainability measurement", "sustainable development", "social accountability", "certification". A total of over 400 articles were found, which were subjected to further examination, whereby the following criteria was applied:

– the characteristics of the social sustainability dimension and in particular the bipolarity of social issues are taken into account
– reference is made to concrete normative frameworks and/or process guidelines and/or management approaches

After application of these criteria, 35 scientific articles remained. These formed the basis for identifying approaches relevant to the management of social issues.

(2) ***Process* 2—*Extraction of core criteria***: In addition, the focus was on extracting sustainability-related core criteria from the relevant approaches. For this purpose, the approaches were evaluated using the qualitative summary content analysis method. The advantage of this evaluation method is based on the systematic and rule-based approach [21]. Each analysis step in the evaluation process can be traced back to well-founded and tested rules. The qualitative content analysis does justice to intersubjective comprehensibility and thus meets the requirements of empirical research. Seven sub-processes (SB) were run through:

(a) *SB* 1—*Determination of the source material*: As already described, the approaches to the management of social topics classified as relevant, formed the material basis of the content analysis. The source material included all the frameworks, process guidelines and management approaches mentioned in the articles.
(b) *SB* 2—*Determination of the analysis units*: In order to increase the precision of content analysis, coding, context, and evaluation units were defined. A keyword was defined as the coding unit, i.e. those nouns that clearly referred to one of the three sustainability dimensions. The context unit consisted of a package of statements on one of the three sustainability dimensions. The evaluation units considered were the individual frameworks, process guidelines and management approaches, which were analyzed one after the other in their entirety.

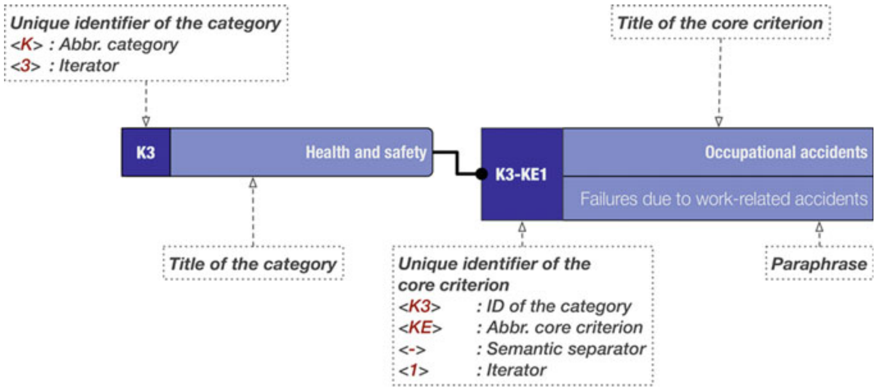

Fig. 1 Basic systematics of the developed interpretation scheme

(c) *SB 3—Paraphrasing of text passages containing content*: First of all, all text components that did not carry content (e.g. repetitive and ornamental phrases) were removed. All text passages with content were paraphrased and then transformed into a short form.
(d) *SB 4—Summary of categories*: The paraphrases could be grouped into categories in a multi-stage procedure. Text passages with similar meanings could be assigned to the corresponding category. With the new identification of text passages that could not be assigned to any of the categories formed, a new category was formed. The execution of this sub-process was repeated until no new categories could be created.
(e) *SB 5—Verification of the category scheme*: The category system extracted from the material was tested against the chosen level of abstraction. The aim was to find out whether the grouping into categories resulted in a significant loss of meaning.
(f) *SB 6—Change of category scheme and coding*: The category system was adapted to reduce loss of meaning and the previously transformed material was coded again.
(g) *SB 7—Interpretation of the category system*: The category system was interpreted with regard to the derivation of core criteria. With the help of the category system, sustainability-related core criteria could be extracted from the relevant approaches and summarized in tabular form. Figure 1 shows the developed interpretation scheme.

Each category received an alphanumeric identifier (e.g. K3) and a title summarizing the contents (e.g. Health and Safety). Following this system, the extracted core criteria could be described unambiguously by an identifier (e.g. K3-KE1), a title (e.g. occupational accidents) and a characteristic paraphrase (e.g. failures due to work-related accidents). An exemplary section of the applied scheme is shown in Fig. 2.

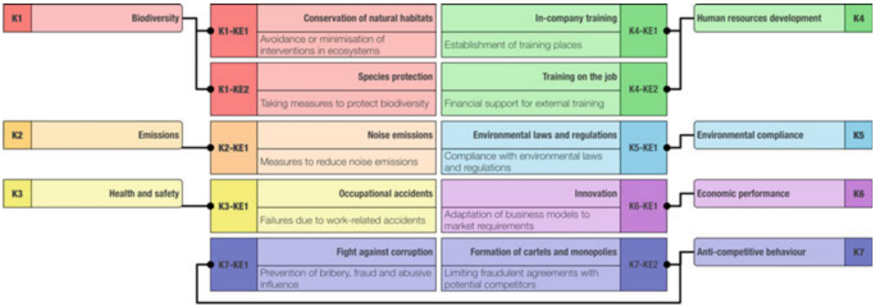

Fig. 2 Exemplary excerpt of the applied interpretation scheme

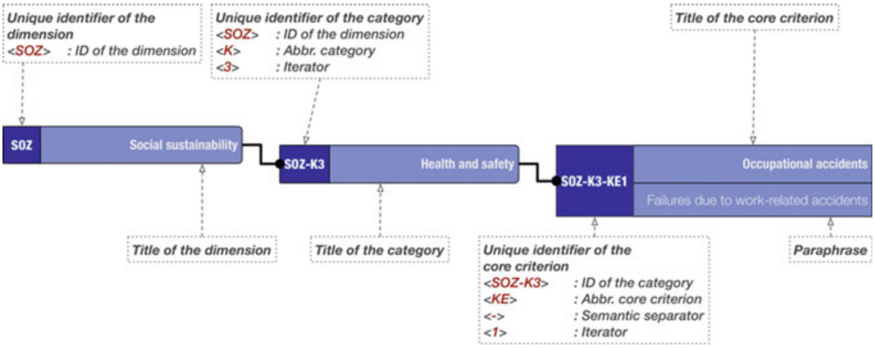

Fig. 3 Exemplary representation of the extended interpretation scheme

(3) ***Process 3—Assignment of core criteria to sustainability dimensions***: In the following, the core criteria of the social sustainability dimension should be separated. For this purpose, the criteria already extracted were each assigned to a sustainability dimension (economic, ecological or social) and then assigned a unique ID. This procedure made it possible to add the dimension to the scheme shown in Fig. 1. Figure 3 shows the interpretation scheme extended by the sustainability dimensions.

The extension of the interpretation scheme follows the basic system shown in Fig. 1, so that each dimension has a unique identifier (e.g. SOZ) and a title summarizing the dimension (e.g. social sustainability). An exemplary section of the extended schema applied is shown in Fig. 4.

(4) ***Process 4—Checking the measurability of the core criteria***: How the core criteria of the social sustainability dimension can be made measurable was the focus of the last process. For this purpose, the interpretation scheme developed in Process 2 and optimized in Process 3 could be applied and the results adjusted for the economic and ecological aspects so that only the core criteria of the social sustainability dimension were mapped. Subsequently, the core criteria could

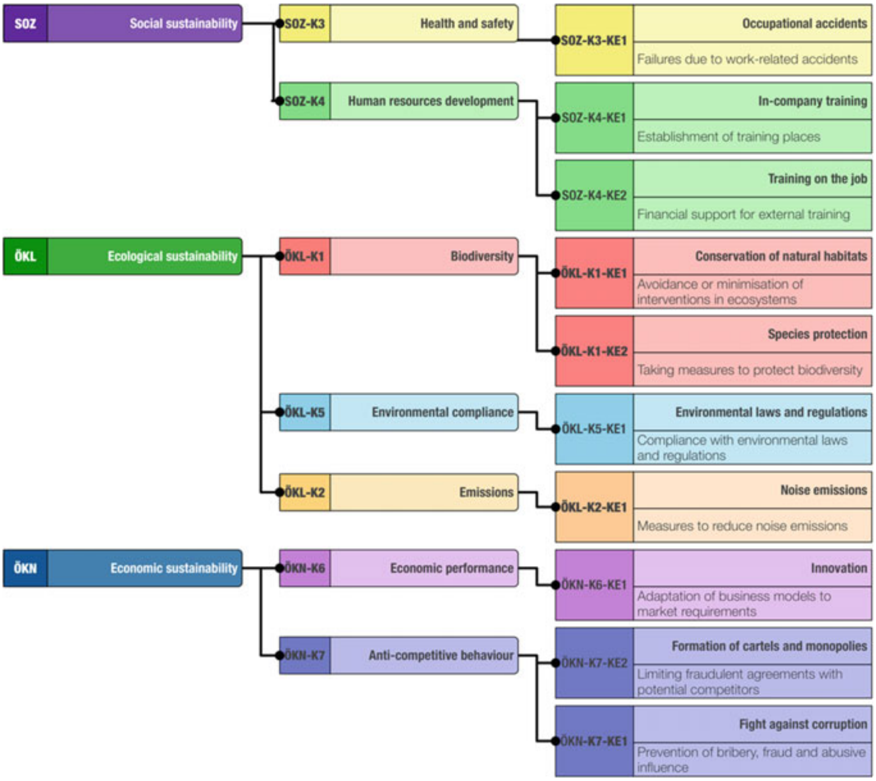

Fig. 4 Section of the applied extended interpretation scheme

be examined with regard to their quantifiability. The following assumptions were made for the test procedure. A criterion is considered quantitative if facts are expressed in key figures. In the case of exclusively described facts, this is a qualitative criterion. A further parameter was added to the scheme. This quantification parameter made it possible to extend the data received to date.

4 Results

The study listed above enabled a total of 600 core criteria to be extracted, which are used to map the sustainability dimensions. After assigning these core criteria to the individual sustainability dimensions, 115 core criteria were identified that reflect the social sustainability dimension. The examination of the criteria with regard to their quantifiability showed that the social dimension of sustainability is primarily qualitatively represented. Of the 115 core criteria, 95 were purely descriptive, i.e. qualitative. A quantitative measurability could be verified for 20 out of 115 core

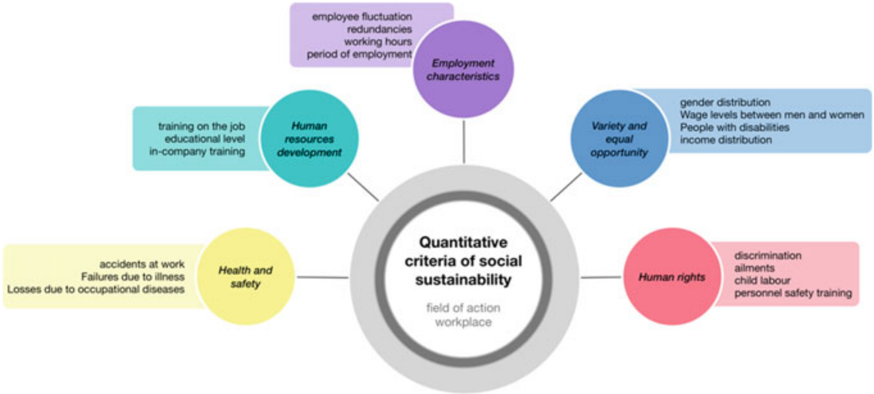

Fig. 5 Categories and extracted core criteria in the area of workplace

criteria, i.e. these criteria were backed up with measurable key figures. This result shows that the social dimension of sustainability is translated into measurable values in existing approaches, but that these remain mainly on a purely descriptive level. In the frameworks, process guidelines and management approaches examined, only 17% of the extracted core criteria were converted into quantifiable values, whereas 83% of the core criteria were presented qualitatively. These results coincide with the findings with regard to the characteristics of social topics in the scientific literature. Dubielzig [16], for example, stated in this context that a quantitative description of numerous social topics, e.g. quality of life, is hardly feasible.

The extracted core criteria show that the social sustainability dimension is only presented quantitatively in the field of workplace action. In the approaches examined, various indicators were sometimes used to quantify the core criteria. Figure 5 shows the categories formed for the field of action workplace, the extracted core criteria as well as exemplary quantitative indicators. Five categories were formed in the field of action "workplace": (1) *Health and safety*, (2) *Human resources development*, (3) *Diversity and equal opportunities*, (4) *Employment characteristics*, (5) *Human rights*.

The *Health and safety* category is described by core criteria aimed at monitoring health and safety risks: accidents at work, absenteeism due to illness, and absenteeism due to occupational diseases. The extracted quantitative core criteria play an important role due to their relevance for the maintenance of operational processes. These core criteria can therefore also have an impact on the economic dimension of sustainability. The core criteria of in-company continuing education, in-company training and the educational level of employees were subsumed under the category of *Human resources development*. Like the core criteria of health and safety, these extracted criteria can also have an impact on the economic dimension. Due to the short half-life of knowledge, well-trained and continuously educated employees can be regarded as an important factor for corporate success. The category *Diversity* *and equal opportunities* summarizes the diversity of the workforce and the equal opportunities of employees. The extracted core criteria of gender distribution, wage levels between men and women, people with disabilities and income distribution was mapped in the approaches examined using quantifiable indicators. The category *Employment characteristics* is characterized by labour policy aspects and contractual agreements and is represented in the approaches examined by the core criteria of employee fluctuation, dismissals, working hours, full-time and part-time employees, length of service and collective agreements. For some of these core criteria, quantification may be required by law, such as recording working time. Other core criteria in this category may have a direct or indirect impact on the company's success and therefore also affect the economic dimension of sustainability. For example, high employee turnover can lead to high replacement costs for employees, high training requirements and the loss of empirical knowledge [22]. Indirect effects result, for example, from the core criterion of redundancies. Cascio [23] stated in this context that a high number of redundancies can reduce employees' commitment and increase their stress levels. The *Human rights* category includes the core criteria of discrimination, child labour and personnel safety training. These core criteria were reflected in the approaches examined using quantifiable indicators. These core criteria can also have an impact on the company's success and thus on the economic sustainability dimension via its image.

5 Discussion

The large number of criteria extracted in the study shows that existing frameworks, process policies and management approaches can be used by companies as a basis for taking stock of social conditions in the processes they monitor. This can be implemented both internally in a gate-to-gate representation and cradle-to-grave over the entire product lifecycle. Furthermore, it became clear that the criteria of social sustainability formulated in the existing approaches could be summarized in a manageable number of categories. This structuring simplifies the development of a category system, but in terms of content it is not always possible to clearly assign facts. For example, an assessment of the working environment according to employees can enable the company to make a statement about the characteristics of employment or the observance of human rights. This shows that core criteria correlate between individual categories. Intracategorial overlaps can also occur. For example, the entitlement to equal pay for equal work irrespective of gender and the survey of income distribution may show a lack of selectivity. Parallel or sequential allocation procedures would have to be applied at this point in analogy to material accounting or impact accounting. In this context, it is up to the companies to ensure that the criteria are adequately defined.

Moreover, in many cases the extracted criteria are closely intertwined with economic determinants, which are also collected in the operational context for other valuation processes. As a result, the catalogue of criteria developed, facilitates an analysis of social sustainability on the basis of existing data or data that can only be marginally supplemented. At the same time, the conflict of objectives between economic decisions and social sustainability requirements becomes particularly apparent at this point. Only those criteria are measured that also have economic effects. The study showed, for example, that the extracted core criteria in the workplace field of action can have a direct or indirect impact on the success of a company. The existing (positive or negative) economic effects of these core criteria are mostly visible and can therefore be understood by management and often depicted in monetary units. Nevertheless, the survey revealed that only a small number of the criteria are quantitatively represented (17%). For the qualitative criteria of social sustainability, there is sometimes a lack of reliable data, which makes it more difficult for companies to objectively map these criteria and to validly measure developments. Conversely, however, this means that only a small proportion of the criteria that are important for social sustainability are taken into account in the sustainability balance sheets of companies. This applies both within and beyond corporate boundaries.

6 Conclusion

The view of Life Cycle Thinking, which has long since been implemented in ecological-economic assessments, poses particular challenges in the context of the expansion to include social stocktaking. The immateriality of the social sustainability dimension and its bipolarity, which refers to both the individual and the societal level, requires a hitherto minor operationalization and a lower systematization compared to the economic and ecological dimension. Based on this initial situation, a scheme was developed to systematically extract criteria of social sustainability from existing frameworks, process guidelines and management approaches. The aim was to support companies in taking greater account of social aspects, both within the company and beyond its boundaries. To this end, existing approaches were examined and a catalogue of criteria was developed that can be operationalized in a company-specific manner and can be flexibly integrated and extended with regard to the standardization of social concerns. However, the majority of the extracted core criteria will continue to be represented qualitatively, so that in order to facilitate accounting in the next step it must be determined how these qualitative criteria can be converted into quantitative criteria. Suitable metrics are needed to objectively map social criteria and thus make them measurable.

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Part II
Mobility

Comparing Carbon Performances of Mobility Services and Private Vehicles from a Life Cycle Perspective

Mara Neef, Tina Dettmer and Liselotte Schebek

Abstract Mobility services are predicted to replace private passenger vehicles to sizeable shares in the short- and middle-term. Although the carbon saving potential of mobility services compared to private vehicles is widely acknowledged, empirical studies are lacking and research designs remain unreplicated. In order to determine common characteristics of studies comparing life cycle carbon emissions of mobility services and passenger vehicles, we conducted a standardized literature review. We showed that current Life Cycle Assessment (LCA)-based approaches in the research field mostly apply two methodological characteristics: (1) person-km (p-km) are used as reference unit to compare carbon performances across transport modes and (2) scenario-analyses are used to deal with the poor data basis and disruptive character of mobility services. Most studies focus on comparing conventionally-powered car sharing vehicles to passenger cars within a one year timeframe in urban areas. Mobility services like ride hailing and pooling as well as alternative power trains remain largely neglected. Policy-makers and customers were found to be the main addressees of case studies. The private sector is least addressed thus showing the need for future research on a mix of mobility services and private vehicles with different power trains on fleet level.

Keywords LCA · CO_2 · PSS · Mobility services

1 Introduction

Passenger transport is causing a sizeable share of carbon emissions which accelerate global warming. In the European Union, 12% of total carbon emissions can be

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attributed to the use phase of passenger cars [1]. Life Cycle Assessment (LCA) studies show that using mobility services instead of private vehicles can be a viable option to reduce carbon emissions related to individual motorized mobility [2–4]. However, [5] point out that only few empirical studies quantifying the carbon performance of mobility services exist and that none of these research designs has been replicated so far. With estimated increased mileages of shared vehicles by 35% in Europe and 46% in China from 2017 until 2030, it is crucial to quantify the effect of using mobility services instead of private cars on carbon emissions and thus on global warming [6].

In order to provide an overview of current approaches that compare CO_2 emissions caused over the life cycle of mobility services and private passenger cars, a standardized literature review is performed to answer the following research question: "What are common characteristics of approaches comparing life cycle carbon emissions of mobility services and passenger cars?". A list of methodological as well as content-related characteristics of the research field will be compiled and existing shortcomings addressed.

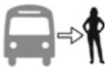
1.1 Life Cycle Assessment and Mobility Services

LCA is used to assess the environmental performance of any product which includes goods and services. The necessary procedures to perform an LCA are defined by the 2006 ISO Standards 14040 [7] and 14044 [8]. In an LCA, consumed resources, emissions as well as impacts on environment and human health are analysed throughout a product's complete life cycle. Within this cradle-to-grave approach, the manufacturing, use and end-of-life (EoL) phases are considered by calculating the resources extracted, the energy consumed and the emissions produced during each phase. As pointed out above, the methodological steps for performing an LCA of a product, e.g. a passenger car, and a service, e.g. a mobility service are the same [9]. Mobility services are referred to as "product service systems" (PSS), i.e. "a mix of tangible products and intangible services designed and combined so that they are jointly capable of fulfilling final customer needs" [10]. Mobility services can further be categorized as use-oriented PSS as the product (vehicle) remains property of the provider [11].

1.2 Passenger Cars Versus Mobility Services

In this study, "passenger cars" refer to Light-Duty Vehicles (LDV) which are purchased by customers. According to the International Organization of Motor Vehicle Manufacturers (OICA), passenger cars have four wheels at least, comprise a maximum of eight seats additional to the driver and are lighter than 3.5 t [12]. Private passenger cars are distinguished from mobility services as the passenger car is bought by a costumer whereas the mobility service vehicle remains the property

Table 1 Characteristics of motorized mobility services

	Direction of service	Individual	Shared
Mobility service		Car sharing (e.g. Car2Go, Drive Now, Flinkster) Car Leasing (e.g. Sixxt)	
		Ride hailing (e.g. Gett, mytaxi, Uber)	Ride pooling (e.g. UberPool, MOIA shuttle)
Power train/fuel		ICE, BEV, PHEV, CNG, Ethanol	
Type of vehicle		Autonomous, non-autonomous	

of the provider. Hence, use patterns of private passenger cars and mobility services differ. Private vehicles are mostly used for only one hour per day whereas the time in which a mobility service vehicle is driven is higher as several people have access to one vehicle [13].

The terms "mobility service" or "Mobility as a Service (MaaS)" refer to digitally connected transport services provided by companies, public institutions or individuals to paying customers which can make owing a private car obsolete [14]. Decreasing numbers of produced vehicles are therefore a main carbon-saving aspect of using mobility services instead of a private vehicle [5].

The scope of this study is motorized mobility services (Table 1). These can be distinguished in individual and shared services, i.e. mobility services for which one individual can determine the exact start and end points versus mobility services which transport several customers with start and endpoints on a similar route. Individual mobility services include car sharing and ride hailing. Members of stationary car sharing programs (also called "car pooling") have access to a vehicle fleet based on a general membership fee plus payments on, e.g., a km-per-trip basis [15]. Free-floating car sharing vehicles are not stationed in specific parking areas as is the case with stationary car sharing but within a wider business area. Also, there are no fixed membership costs and booking in advance is unnecessary [16]. Car leasing is another individual mobility service as the costumer is not owning the vehicle but paying on a kilometre basis. The difference to car sharing is that during the leasing time no other customer can access the vehicle. Therefore, the use pattern of private and leased vehicles does not differ as [4] pointed out.

In contrast to car sharing programs, ride hailing (also called "ride sourcing") works similar to ordering a taxi: customers order and pay the mobility service via app and are picked up at the indicated location. This represents a change in the direction of service: car sharing customers have to locate and reach the vehicle by themselves whereas ride hailing customers order the vehicle to the desired location. The degree of servitization is thus higher for ride hailing programs.

Ride pooling is an example for a shared mobility service, i.e. several customers determine the direction of travel of a ride pooling vehicle. Customers plan start and end points of their travels via app. Algorithms calculate which customers are picked up by a specific vehicle based on the similarity of their journeys [17]. As in the case of ride hailing, the service vehicle is locating the customer.

Other than car sharing and car leasing, ride hailing and ride pooling lead to temporary "empty travel" of service vehicles. Distances driven in order to reach the customer when no other passengers are on board are empty travels as fuel or electricity is used to travel but no mobility is provided.

Mobility services are offered by specialized companies like Uber (uber.com) which offers ride hailing and pooling as well as firms expanding their business portfolio such as the German train provider Deutsche Bahn with its car sharing program "Flinkster" (flinkster.de). Several automotive Original Equipment Manufacturers (OEM) also started offering mobility services. Ride hailing programs offered by OEMs include: "my taxi" by Daimler and "Gett" in which Volkwagen invested [5, 18, 19] provide an extensive overview of car sharing programs offered by OEMs. Examples include "Car2go" by Daimler and "DriveNow" by BMW [20, 21]. "MOIA shuttle" is the ride pooling program offered by Volkswagen [18]. According to [22], reasons for OEMs to invest in mobility services include the prospect of opening up a new market and selling environmentally-friendly mobility. The carbon-saving potential of OEMs selling mobility services instead of vehicles is, however, not yet quantified [5]. Motorized mobility services can further be distinguished by their powertrains like Internal Combustion Engines (ICE), (Plug-in) Hybrid Electric Vehicles ((P)HEV), Battery Electric Vehicles (BEV) and fuels used like Compressed Natural Gas (CNG) and Ethanol. In the future, an additional category will be autonomous and non-autonomous mobility services. In the case of autonomous or automated vehicles, the occupancy rate can be increased as an additional passenger can occupy the former driver's seat.

The paper is structured as follows. In Chap. 2, we describe the applied literature review method. In Chap. 3 the results of the review are presented and discussed in the subsequent chapter. A conclusion with final recommendations and outlook is provided in the last section.

2 Methodology

In a prior screening of the literature, no review article was found that specifically compares characteristics of LCA-based approaches analyzing the carbon performance of mobility services and private passenger cars. Therefore, this article aims at providing an overview of methods proposed so far. In a systematic literature review a clearly defined process is set up in order to generate replicable results [23]. Based on a hybrid approach of [24] and [25] as presented in [26], we set up three steps to structure this review. First, we identify the research question, adequate search terms and decide on databanks used. Second, general screening criteria as well as specific inclusion and exclusion criteria are set up.

2.1 Research Question, Databases, Search Terms

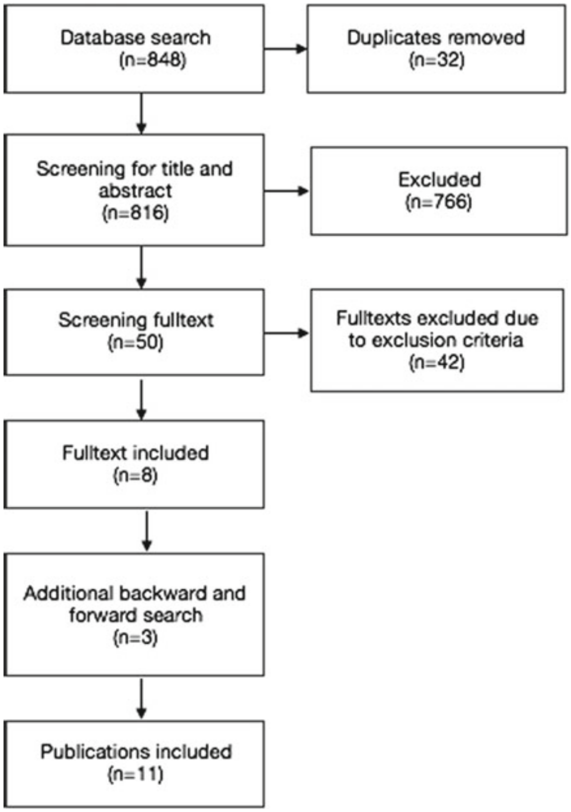
The research question of this paper is broken down into sub-questions to structure the analysis. Concerning characteristics of the assessed mobility services, we want to find out (a) what kind of mobility service is subject of the study and (b) what power trains resp. fuels used are attributed to the service and private vehicle (see Table 1). Like this, conclusions drawn from the studies are set into perspective as different mobility services and power train fleet compositions result in different statements concerning carbon performances. For (a) and (b) the following categories were set up: (a) mobility service: (non-)/autonomous car sharing/car leasing/ride hailing/ride pooling, (b) power trains/fuels: ICE/(P)HEV/BEV/CNG/Ethanol. Concerning methodological characteristics, we distinguish between (c) the geographic scope, (d) time horizon and (e) audience of the studies. For (c)-(d) the following categories were set up: (c) geographic scope: city/country/rural area, (d) time horizon: <1 year/1 year/vehicle lifetime. Possible addressees of the studies are persons or organizations interested in gaining knowledge about the carbon-saving potential of mobility services compared to private vehicles. Next to the money-saving potential of using mobility services instead of owning a vehicle, climate-conscious consumers could be interested in mobility services' carbon performance. Also, policy-makers deciding whether infrastructure for mobility services should be subsidised might have an interest in, e.g., which mix of power trains in a mobility service fleet is most carbon-efficient. The private sector as the provider of mobility services has an interest in estimating the carbon efficiency of their vehicles and fleet. Finally, methodological publications are directed at academia. Therefore, the following categories are set up for (e) audience: customers/policy-makers/private sector/academia. We further collect information on (f) life cycle phases covered and (g) reference units used to assess carbon efficiencies of the respective mobility services. As for (f), the three life cycle phases or a combination of them represent the respective categories: (f) life cycle phases: manufacturing + use + EoL/combination of phases. The reference units for assessing carbon performances of vehicles can, depending of the focus of the study, either be vehicle-kilometres (v-km) resp. vehicle-miles (v-m) or person-kilometres (p-km) resp. person-miles (p-m). In the case of the reference unit of p-km, occupancy grades and the distance driven are taken into account. Categories for (g) reference unit are thus: v-km/p-km/v-km + p-km. In case p-km are used in (g), we collect information on (h) assumed occupancy grades. Here, no categories are predefined.

Furthermore, (i) the type of LCA-approach is distinguished. Studies can pursue an attributional or consequential LCA-approach as well as a combination resp. elements of the two. Attributional LCA (or accounting/book-keeping LCA) is used to describe a product's environmental impact over the whole life cycle. The product is modelled in its current or planned state. Therefore, specific data for manufacturing, operation and EoL phases can be used. Attributional modelling is also used for product comparisons. Consequential LCA is not focusing on a specific state but is change-oriented. It assesses in how far decisions taken in a foreground system affect the market or consumer behaviour in the background system [27]. The so-called rebound effect can be included in consequential LCA studies [28, 29]. It describes direct and indirect effects of products or services on individual or collective consumption or production patterns [30]. Possible effects of mobility services on customers' mobility patterns and demands could be crucial when assessing the carbon performance of mobility services. Therefore, the categories for (h) type of LCA are: accounting/consequential/mix of elements. Finally, (i) mentioned challenges when assessing carbon emissions of mobility services are examined. Here, no categories are predefined. Chosen databanks include Elsevier (sciencedirect.com), Scopus (scopus.com) and Web of Science (webofknowledge.com). For screening title, abstract and keywords, the following search term was used: accounting OR LCA OR "life cycle assessment" AND CO2 OR carbon OR GHG OR "greenhouse gas" AND "mobility service" OR "car sharing" OR PSS OR "product service system".

2.2 *Screening Criteria, Inclusion and Exclusion Criteria*

In order to generate an exhaustive overview of the literature published on the topic, we included publications without restrictions of publishing date in English language. No further quality criteria as to preferring higher ranked journals were applied. Case studies as well as review articles were included. First, retrieved publications were screened for relevance by sighting titles and abstracts. Based on the research question, the following inclusion criteria were exercised on the publications left for analysis after primary screening: (1) The study comprises a quantitative life-cycle based approach for modelling carbon emissions of mobility services and private passenger vehicles. As inclusion criteria (1) yielded a very small amount of eligible papers, it was reformulated. Although the ultimate goal of this work is to review methodologies including the whole life cycle, the crucial life cycle phase for the overall carbon performance of private and mobility service vehicles is the use phase. During the use phase, most carbon emissions of ICE-powered vehicles are produced [31]. Also, in the use phase, the difference between product (vehicle) and PSS (mobility service) becomes apparent as occupancy grades can differ among the two. The reformulated inclusion criteria is: (1a) The study comprises a quantitative approach for at least the use phase of mobility services and private passenger vehicles. (2) The study deals with any of the following mobility services: (non-)/autonomous car sharing/car leasing/ride hailing/ride pooling. (3) The mobility service dealt with is powered by the following power trains resp. fuels: ICE/PHEV/BEV/CNG. (4) The study deals with mobility services as an example within PSS carbon emissions modelling.

Fig. 1 Workflow diagram with quantitative results



3 Results

By using our search term in the selected databases, 848 hits were yielded. After 32 duplicates were removed, titles and abstracts of 816 publications were screened. After excluding 766 papers, the full texts of 50 selected publications were analysed by applying inclusion and exclusion criteria. As this procedure resulted in only eight eligible publications, an additional forward and backward search of publications cited in the selected papers was performed as proposed by [32]. This procedure resulted in three additional relevant publications. The complete work flow according to [33] with quantitative results is shown in Fig. 1.

3.1 Bibliographic Analysis

The selected papers are published between 2005 and 2017; 82% in peer-reviewed journals. Two are publications of university institutes (e.g. [34]). Nearly half of the papers (45%) are published in 2016 or later. 82% of publications represent case studies whereas only one publication develops the classic LCA approach further [35] and one offers a literature review on accounting methodologies of different PSS [9].

3.2 Characteristics of Assessed Mobility Services

Speaking in the classification of mobility services presented above (Table 1), 91% of papers use car sharing as at least one of their mobility services. In the case of [35], shared autonomous vehicles (SAVs) are subject of the study. As these SAVs would be ordered to a specific location, this service can be categorized as autonomous ride hailing (Table 1). Only [4] and [36] include additional mobility services such as car leasing and ride hailing. Most authors include ICE-powered vehicles in their studies (91%). 28% include additional hybrid vehicles and 18% analyse BEVs. Only [4] include ethanol and CNG-powered vehicles. As [9] is a methodological publication, no case study is provided.

3.3 Methodological Characteristics

The geographic scope of the case studies is prevalently a specific city (36%) (e.g. [3]), respectively a model urban area (e.g. [37]) or a country (28%) (e.g. [2]). Only [4] analyse mobility services within a more rural setting ("small towns"). 55% of the studies used one year as their reference timeframe. One paper each used 100 days [35], respectively a vehicle's lifetime of 119,780 miles [38] and an undefined vehicle's lifetime [37]. [36] and [9] state no specific modelling timeframe.

Addressees of studies are mostly not explicitly stated so that criteria had to be set up to categorize different audiences. In case individual carbon savings by joining a mobility service were stated, "customers" were chosen as an audience. In case extrapolations of carbon savings by setting up a mobility service program on city, country or region level were stated, "policy-makers" were chosen as an audience. In case information was provided on carbon efficiencies on mobility service fleet level, "private sector" was chosen. In the case of methodological publications, "academia" was set as audience. As most publications fit several of the categories, several audience types were selected. Policy-makers are addressed in 82% of publications followed by customers (64%). Analyses of carbon performance on fleet level was provided in 45% of the studies. [9] address scholars only.

As the database search was designed for LCA-based accounting methods, seven out of the eleven publications included the whole life cycle of mobility services in their analyses. Another three only accounted for carbon emissions of the use phase (e.g. [15]) and one focused on manufacturing and use phase [35].

The majority of authors chose p-km as the reference unit for their studies. Additionally, [9] underline that generally the functional unit of LCA approaches should display the function offered by the assessed product or service. In the context of mobility services, this refers to relating modelling results to p-km rather than v-km in order to be able to compare capacity utilizations across transport modes. This adds up to 64% of publications choosing p-km as reference unit. 28% measure carbon performances of service and private vehicles in v-km and 9% use v-km as well as p-km.

Assumed occupancy rates of car sharing vehicles range from 1.39 passengers [2] to 4.59 passengers [38]. Four publications follow a consequential approach to varying degrees. [39] and [15] subtract avoided CO_2 emissions by using a private vehicle from CO_2 emissions caused by using car sharing, the so-called “counterfactual”. [37] include avoided CO_2 emissions from less urban parking space needed for car sharing vehicles in relation to private vehicles. Both [37] and [34] include impacts of using car sharing on other transport modes such as rail, bus and aircraft. [34] further include indirect economy-wide effects to analyse rebound effects caused by introducing car sharing programs. The money saved from joining car sharing is expected to be spent equally across sectors other than transport. [9] refer to rebound effects as a decisive parameter in assessing the environmental performance of PSS.

Challenges mentioned when modelling emissions caused by car sharing programs include the variance in available information regarding, e.g., private vehicles replaced by shared vehicles [39]. For example, [39] assume that one shared vehicle replaces six private cars in Lisbon whereas [37] assume a range of nine to 23 private vehicles being replaced in U.S. urban areas. [2] point to the difficulty of establishing causality between joining car sharing and changing mobility behaviour. According to them, numerous other factors like, e.g., starting a family can influence personal mobility demand and modes of transport chosen. [37] state that due to data insecurity it is demanding to include economy-wide rebound effects. [34] underline this problem by acknowledging that rebound effect calculations are based on assumptions only. Finally, [9] point out the general difficulty of evaluating the environmental performance of PSS ex ante, i.e. before their market introduction. As PSS can present a radical change of user experience in a future society whose characteristics are yet unknown, the reliability of such studies is often limited. Furthermore, due to the variance in data describing the impact of car sharing on individual mobility behaviour, 64% of the selected studies use a scenario approach to display a range of possible results and conclusions. [9] also recommend using a scenario approach when analysing environmental impacts of PSS.

4 Discussion

Although ride hailing and ride pooling serve increasing shares of urban mobility demand, car sharing is the main mobility service analysed in the selected studies. This might be due to the rather long time span car sharing programs have been available to customers. Accordingly, all primary data used in the selected publications was derived from car sharing programs only. Still, as pointed out before, the lack of reliable empirical data is a challenge for the carbon performance analyses of all mobility services [5]. This problem also becomes apparent when analysing the differences in values used for p-km provided by car sharing vehicles and occupancy rates. [4] use 17,500 p-km per shared vehicle per year whereas [2] use 27,750 p-km per shared vehicle per year. The difference is due to different assumed personal mobility budgets of 1850 and 3500 p-km/year, resp., different numbers of people using a shared vehicle (5 and 15 people, respectively). Since no common values for occupancy grades and v-km are used to estimate the amount of personal mobility provided by a shared vehicle, results of different studies cannot be compared. The difference in these data inputs cannot only be attributed to varying geographic locations. As [37] point out, LCA studies dealing with car sharing mostly do not rely on primary data but on estimated data from similar studies. Like this, biases can be manifested throughout different publications. For example, [34, 39] and [2] partly rely on data collected in North America although they analyse European mobility behaviour.

A possibility would be to use a set of common values such as occupancy grades for passenger cars as reported by governmental institutions (e.g. 1.45 passengers per vehicle in the EU and 1.59 passengers per vehicle in the U.S. [40, 41] and a set vehicle lifetime mileage of, e.g., 150,000 v-km as reported by OEMs [42, 43]. Still, for occupancy grades and lifetime kilometrage of all mobility services, more empirical studies in different geographic locations are needed.

Current studies mostly focus on comparing carbon performances of ICE-powered private and shared vehicles within a timeframe of one year in a specific city or country. The audience for these studies was shown to be prevalently policy-makers and customers. The private sector, i.e. companies deciding on mobility service fleets' size and power train mix are least addressed among the case studies. With projected increasing shares of mobility services replacing private vehicles, the private sector, i.e. analyses of life cycle carbon emissions of service fleets composed of different kinds of services and power trains, need to be addressed more. Like this, advice on most carbon-efficient mobility service fleets could be provided.

The methodological analysis further showed that authors of the majority of studies use (or recommend to use) the same parameters to compare carbon saving potentials of private passenger vehicles and mobility service vehicles. In most publications, a holistic approach modelling CO_2 emissions over the whole life cycle of private and service vehicles is pursued. Although the difference between private and service vehicles becomes most apparent during the use phase, including carbon emissions caused in manufacturing and EoL phases will become more important with higher shares of electrified power trains in the fleets. And, according to [6], the rise of mobility concepts will be accompanied by a rise of electrified and autonomous vehicles.

The reference unit most adequate to compare life cycle carbon emissions of different passenger transport modes was shown to be 'kg CO_2 per p-km'. Like this, not the vehicle itself but the carbon performance per transported passenger is assessed thus taking into account varying occupancy grades of different transport modes. The major concern of data availability and quality was met with scenario-building. Therefore, selecting appropriate scenarios is key. Additional sensitivity analyses can increase reliability of results as performed by [35]. Applying a consequential approach by including e.g. a rebound effect further increases the demands on data quality [9]. Also, the variety of consequential elements like avoided emissions from decreased parking space or economy-wide rebound effects make a comparison of results impossible. Though [2] underline the difficulty of determining causal relations of individual mobility behaviour and chosen transport mode, an effect of mobility services on overall mobility demand is worth looking into. Depending on the magnitude of, e.g., increased mobility demand due to affordable and comfortable mobility services, the carbon saving potential of mobility services compared to private vehicles could be diminished. Finally, for ride hailing and pooling, additional or "empty" travels to reach the next customer should be included [35].

5 Conclusion

This literature review showed that current methodologies comparing carbon performances of private passenger cars and mobility services rely on some common parameters. In the majority of publications, a holistic life cycle approach by taking into account manufacturing, use and EoL phases of vehicles is pursued. Comparative carbon performances of service and private vehicles are analysed with 'p-km' as reference unit to take into account varying occupancy grades across transport modes. Challenges with data quality and reliability are met with scenario-building to account for a range of future developments. Though different scenarios, consequential approaches and input parameters like occupancy grades, total vehicle kilometrage and fleet mixes make comparing results of studies impossible. Regarding the content, the focus lies mainly on comparing gasoline or Diesel-powered car sharing vehicles with private vehicles. Like this, probably higher carbon saving potentials of mobility services with higher occupancy grades like ride pooling remain so far underrepresented. Only one publication analysed ride hailing vehicles pointing out the need to account for so-called "empty travels" induced by pick-up services of the mobility service. This additional vehicle kilometrage and thus carbon emissions must also be included in the analysis of ride pooling vehicles. With projected increasing shares of mobility services, possible feedback-loops of affordable mobility services on mobility demand should be analysed. Finally, the private sector was shown to be least addressed in case studies thus showing the need of analysing life cycle carbon performances of mobility services on fleet level in the future.

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Sustainability Impacts of Mobility as a Service: A Scoping Study for Technology Assessment

Rikka Wittstock and Frank Teuteberg

Abstract The potential for positive sustainability impacts of Mobility as a Service schemes is frequently mentioned in both scientific literature and public media, although a systematic evaluation of potential impacts is lacking thus far. In preparation of an in-depth technology assessment, we conduct a scoping study aimed at achieving a better conceptualization of what core elements constitute Mobility as a Service, what risks and opportunities are associated with this concept and how these may be further analyzed as part of a technology assessment project. Reviewing a total of 95 sources from academic literature as well as grey literature and media reports, we provide a synthesis of the core elements of Mobility as a Service schemes, develop hypotheses on the risks and opportunities involved and propose a framework for further assessment of the associated sustainability impacts.

Keywords Technology assessment · Mobility as a service · Literature review · Scoping study · Sustainability

1 Introduction

Closely interlinked with social and societal processes, innovations cannot be viewed as isolated, purely technological progress but should be considered within their unique context of sociotechnical development. With the aim of making strategic decisions in both politics and industry, a careful consideration of an innovative technology's risk potentials as well as the resulting opportunities for society therefore is imperative [1].

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With regards to "Mobility as a Service" (MaaS), the potential positive societal impacts of a further diffusion of this concept are heralded by both media and scholars, including, for example, potentials for a reduction of the environmental damage caused by urban traffic, the extension of mobility services available to citizens and the creation of new business opportunities for the transportation industry. Such positive impacts appear to be accepted as a logical consequence of the MaaS concept, however, a systematic assessment of its possible impacts is lacking thus far.

Currently, "Mobility as a Service" constitutes a fuzzy and ill-defined concept, with a large part of current MaaS research focusing indeed on conceptualizations of MaaS and its associated terms. In particular, the implications for society, environment and business deriving from a further diffusion of this concept remain largely unknown. As of today, very few pilot studies have been performed and only individual MaaS projects have actually been implemented in practice. Empirical evidence of the concept's real-world impacts is therefore lacking. Current studies hence list a lack of applicable norms and standards, insufficient data security, a lack of legal bases and insecurity concerning the economic value as major reasons for the present hesitation of businesses and local authorities to invest in MaaS.

Against this background, the following research questions are to be addressed as part of this paper:

1. Which concepts, core characteristics and elements are currently associated with the term "Mobility as a Service"?
2. What opportunities and risks are linked to the term "Mobility as a Service"?
3. Which further objectives are relevant for the project design, and which methods are suitable for addressing these?

In order to address these research questions we conduct a scoping study that is to function as a basis for the ensuing technology assessment. Synthesizing and analyzing a wide range of academic and non-academic material to provide greater conceptual clarity concerning the situational context of MaaS, we seek to establish core elements of MaaS schemes, identify opportunities and risks involved and derive suitable methods for a further in-depth analysis.

This paper is structured as follows: In Sect. 2, we explain the technology assessment methodology and in particular, the chosen approach for addressing the research questions identified above. In Sect. 3.1, we expand upon the concept of "Mobility as a Service", outlining the variety of current research streams within this field as well as introducing core elements of this mobility model. Section 3.2 presents the identified risks and opportunities of a further MaaS diffusion. In Sect. 4, these hypotheses are discussed with regards to the relevance of an in-depth technology assessment project and a project design draft is proposed. Conclusions are drawn in Sect. 5.

2 Methodology

Technology assessment (TA) was developed in the United States (US) in the 1960s with the aim of providing the US Congress with a better understanding of what could be the economic, political, ethical and other consequences of the introduction of a new technology into society [2]. TA hence constitutes a scientific assessment approach by which the societal benefits and detriments of a particular technology are weighed against each other [3]. As in Life Cycle Assessment (LCA), environmental impacts of new technologies or of the expansion of existing ones are one of the key focus areas of TA [4]. However, TA considers these environmental impacts in the context of much broader societal impacts, and in addition to ecological consequences explicitly includes economic, social, legal, ethical and technological aspects [3].

Since TA is generally used to assess the consequences of the introduction of relatively novel technologies, the related impacts typically lie in the future and cannot be observed empirically. Instead, TA minimizes the existing uncertainties with regards to implementation of the technology by collating all available information in order to produce the best-possible basis for decision-making [1]. In order to achieve a holistic understanding of the developmental context of a specific technology the combined application of methods from a range of different academic disciplines is necessary [1]. The general process followed as part of a TA project is pictured in Fig. 1.

A scoping study, aimed at synthesizing and analyzing a wide range of research and non-research material to provide greater conceptual clarity of the reviewed technology's situational context, forms the central starting point of any TA project. In this step, a clear understanding of the current and, as far as possible, future problem context of the respective technology is formed by analyzing the scientific and public

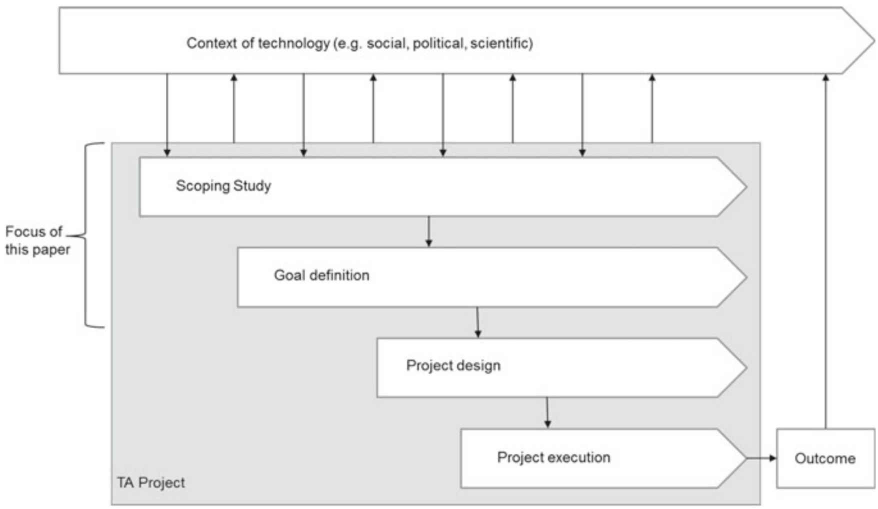

Fig. 1 Process of a technology assessment project. Adapted from [3]

debate on this topic. Based on the identified issues, goal definitions and applicable methods are derived for the ensuing TA project [1, 3].

With the aim of achieving a better conceptualization of what core elements constitute Mobility as a Service, what risks and opportunities are associated with this concept and how these may be further analyzed as part of our technology assessment project, we review academic literature, grey literature and public media statements in the following sections.

2.1 Related Work

As part of our literature search applying the keywords "*Mobility as a Service*" *AND* "*technology assessment*" to the search engines Google and Google Scholar in both English and German language, only 4 related publications dealing foremost with the impacts of MaaS could be identified in June 2018.

First of all, Rantasila [5] described a number of MaaS pilot projects and trials and hypothesized how a further diffusion of MaaS may impact on land use in the Finnish context.

Scholars of the Swedish Chalmers University published a research report on MaaS in 2017 [6] portraying their developed assessment framework for evaluating positive and negative outcomes of MaaS on an individual, organisational and societal level. Although empirical information was either taken from the Gothenburg UbiGO trial only or unavailable for a number of the proposed key performance indicators, the authors suggest a generally beneficial change from a further diffusion of MaaS.

Arnold et al. [7] as part of a project examining the potential of a range of mobility services for reducing the reliance on private vehicles, attempt to assess the diffusion potential of Mobility as a Service. Based on a literature study, the authors review a number of transition pathways and scenarios describing how the concept, business model and involved actors of MaaS may evolve over time.

Finally, Niggebrugge et al. [8] propose a software-assisted method for evaluating the sustainability impacts of Mobility as a Service. Focusing on a case study of Amsterdam, they propose an assessment framework from the end-users' and mobility providers' perspective. Special attention is given to interrelationships and dynamics between the different impacts, for example, what immediate, enabling and systemic effects are caused by the proposed higher flexibility of mobility means.

2.2 Literature Analysis

In order to develop a well-founded definition of the term "Mobility as a Service" and derive expected risks and opportunities within this situational context, we performed a literature analysis following the approach given by Webster and Watson [9], focusing on academic journals and conference proceedings to begin with. In

June 2018, the keywords "*Mobility as a Service*" *OR* "*MaaS*" (in both English and German language) were applied to title, abstract and keywords within the literature databases Google Scholar, Science Direct and AiSeL. With the aim of receiving a high number of matching results, no further restrictions were applied regarding, for example, date of publication or methods used. Following a review of the identified papers' abstracts or, in cases where this was insufficient for determining a paper's relevance, the full content, 37 publications were considered relevant and reviewed in further detail as part of the situational analysis.

2.3 Grey Literature and Media Resonance

Since Mobility as a Service constitutes a relatively new phenomenon, few scientific publications exist that deal with this topic and its associated risks and opportunities. For this reason, we extended the literature analysis to include so-called grey literature (i.e. project reports, position papers, white papers and institutional reports). In addition, websites of companies and trade associations, blogs as well as trade and public media reports dealing with MaaS were considered. This approach is especially useful since a major aim of the situational analysis is to examine the public understanding of and response towards the technology that is to be assessed. For this purpose, an internet search using the keywords "*Mobility as a Service*" *OR* "*MaaS*" (in both English and German language) was conducted in July 2018. Considering the first 150 hits of the respective search engines, this approach delivered 58 results, 36 of which were classed as grey literature and 22 web-based and media-related publications.

The final sample of relevant literature sources reviewed in detail as part of the situational analysis therefore consists of 37 scholarly publications, 22 web-based and media-related publications and 36 publications that are defined as grey literature. A full list of the analyzed publications can be retrieved from https://bit.ly/2LoN0Bh.

Based on the research questions defined in Sect. 1, we follow the approach for qualitative content analysis as given by Kuckartz [10] in order to derive a well-founded overview of the core elements associated with the term "Mobility as a Service" as well as hypotheses regarding the associated societal risks and opportunities. The results of this analysis are summarized in Sect. 3.

3 Results

3.1 Definitions and Core Elements

Mobility as a Service constitutes a very recent mobility concept. What is or is not MaaS remains a highly contested subject, with many sources offering conflicting definitions or describing divergent core characteristics. As empirical observations and structured analyses of the few implemented MaaS schemes are yet to be conducted, current research streams published in scientific literature focus mainly on the following aspects:

- Theory development and characterization (cf. [11, 12])
- Business models (cf. [13–16])
- Market potential and consumer acceptance (cf. [17])
- Reviews of pilot projects (cf. [18–20])
- Exploration of policies and developments enabling MaaS diffusion (cf. [13, 14, 21, 22]).

Alluding to concepts of the information and communication technology community (e.g. software-as-a-service or platform-as-a-service (cf. [23, 24]) the term was first coined by Hietanen [25] who envisioned a mobility concept "in which a customer's major transportation needs are met over one interface and are offered by a service provider". In this interpretation, MaaS is understood to combine different transport modes within one tailored mobility package offering, analogous to a monthly phone subscription, and comprises the following key aspects: service bundling, cooperation and interconnection between different transportation modes and providers that is based on consumer needs [11].

Several publications have extended upon this interpretation and emphasized different aspects. For example, Holmberg et al. [26] focus on the role of service bundling and subscription, arguing that MaaS integrates a wide range of transport services, including both traditional services, such as public transport, and options made possible through more recent technology developments, such as internet-accessed peer-to-peer transportation. Similarly, Sochor et al. [19] as well as Sarisini et al. [16] describe MaaS as a bundling of both public and private transportation options that are offered by one service provider, with information, booking and payment of services being handled via mobile applications.

Others, including Belletti and Bayen [27] and Callegati et al. [28], propose a much wider scope and encompass a range of on-demand and dynamic transportation services under the term Mobility as a Service.

The central function of internet and other technologies is highlighted, for example, by Karmagianni and Matyas [12], Kriukelyte [29], Yadav et al. [30], Matyas and Karmagianni [31] 2018, Sarasini et al. [16], and Li and Voege [22] who emphasize a level of integration that is to enable booking and paying for services via a unified, digital interface. This user-centric approach, which interprets travel, and especially the time and energy spent searching for and booking transportation, as a disutility to consumers constitutes a further key aspect of MaaS that is addressed, for example, by Mulley [32], Hensher [15], Jittrapirom [11].

In contrast, Sarasini et al. [16] argue that there is little sense in trying to define Mobility as a Service at this point in time since the concept entails a vast variety of innovative solutions, all of which are currently in a state of fluidity.

Interestingly, sustainability is seen as a core characteristic of MaaS by a notable number of publications [11]. For example, Sarasini et al. [16] mention that MaaS

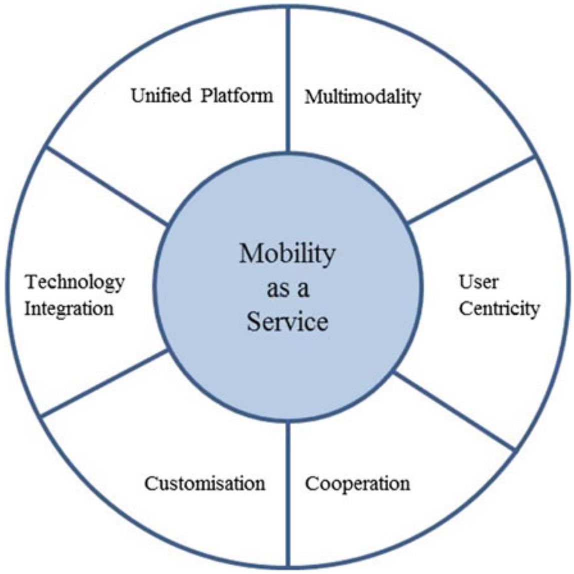

Fig. 2 Core elements of mobility as a service schemes

has been posed as a strategy for delivering sustainability gains in terms of reduced congestion, while Gould et al. [33] and Sochor et al. [34] envision an opportunity for reorganizing the mobility system to reduce dependence on private vehicles.

Within this paper, we adopt a relatively broad interpretation of Mobility as a Service. Following Smith et al. [35] we view MaaS as a bundled offering of public and private transportation services that allows users to travel from one place to another using different transport modes. We propose the following core elements as derived from our literature review (Fig. 2):

Multimodality
Multimodal transportation services that allow users to select from a variety of different transport options is one of the major goals of MaaS schemes. The discussion currently focuses on urban transportation, including for example public transport, car-sharing, bike-sharing, ride-hailing, and on-demand services; however, the inclusion of long-distance options, such as rail or air travel, is envisioned in some cases.

User Centricity
Instead of buying the means to transport (i.e. usually a car), users of MaaS buy mobility. Meeting users' transportation needs and allowing them to get from one place to another seamlessly is therefore at the core of the MaaS concept. This includes not only the integration of on-demand services, but also the entire customer experience before and after the actual act of travelling. Depending on the individual scheme the latter aspect can be more or less advanced, with services ranging from the possibility to find travel information to one-stop options offering mobility packages and even additional services, such as goods delivery.

Customisation
Catering directly to user needs, MaaS schemes generally include options for customisation and personalisation, such as offering tailor-made solution based on individual preferences, booking history or volume of travel.

Cooperation
A key characteristic differentiating MaaS from other mobility models is the fact that such schemes require the integration and cooperation of actors from a variety of domains, including public and private transport companies, local authorities, platform providers, data management companies and payment service providers.

Technology Integration
MaaS business models require the integration of a range of technologies, including for example telecommunication services, e-payment systems, data management and integrated infrastructure data.

Unified Platform
The service is accessible via a digital platform (generally a smartphone app supplemented by a website) that integrates all offered services and covers the entire customer experience from gathering travel information, to booking, ticketing and paying for services. Subscription-based platforms typically require upfront registration.

3.2 Risks and Opportunities

Following the conceptualization and identification of core elements associated with the term Mobility as a Service, we reviewed the literature sample to identify risks and opportunities raised in relation to this mobility model. Since it is our aim to assess potential beneficial and detrimental impacts on the wider society, we divide the identified risks and opportunities into environmental (En), social (So) and economic (Ec) impact. In order to facilitate the later in-depth of analysis of these risks and opportunities as part of the future technology assessment project, the individual statements were summarized as hypotheses in a second step. Tables 1 and 2 list the risks and opportunities brought up in the analyzed literature sample as well as the developed hypotheses.

4 Discussion and Outlook

The scoping study, seeking to improve understanding of the core elements that constitute Mobility as a Service as well as the risks and opportunities associated with this scheme, demonstrates that the concept is still in its early stages of development with little empirical evidence to support its anticipated benefits. Although the topic has sparked a high level of interest by researchers, businesses and media alike, a

Table 1 Opportunities identified in literature analysis

Opportunities as stated in literature	Resulting hypotheses	Impact category		
		En	So	Ec
Reduction of car ownership, reduction of congestion, exchange of outdated technology, higher efficiency of public transport, improved accessibility of public transport, on-demand scheduling, reduction of idle time, pollution and emission reduction (unspecified)	H1.1: The higher the level of adoption of MaaS services, the lower are emissions of greenhouse gases (CO_2 and equivalents)	x		
Reduction of car ownership, reduction of congestion, exchange of outdated technology, higher efficiency of public transport, improved accessibility of public transport, on-demand scheduling, pollution and emission reduction (unspecified)	H1.2: The higher the level of adoption of MaaS services, the lower are emissions of air pollutants (PM, VOC, NO_x, CO, SO_2)	x	x	
Reduction of inefficiencies, reduction of dependence on subsidies, customisation, access to customer data, reduction of idle time, higher competitiveness (unspecified)	H1.3: The more a business is involved in the creation of MaaS services, the higher is its later competitiveness			x
Improved quality of planning, higher level of integration, higher level of collaboration	H1.4: The more a business is invested in MaaS services, the higher is the level of collaboration with related businesses			x
Improved accessibility of transportation, on-demand scheduling, lower individual cost, interaction and communication, transparency	H1.5: The higher the level of MaaS diffusion, the higher is the social equity experienced by citizens		x	

Table 2 Risks identified in literature analysis

Risks as stated in literature	Resulting hypotheses	Impact category		
		En	So	Ec
Shift away from public transport, incentive for on-demand services	H2.1: The higher the level of adoption of MaaS services, the higher is the share of non-ecological transport options in the modal split	x		
Shift away from public transport, higher number of journeys	H2.2: The lower the individual cost of mobility, the higher are psychological and financial rebound effects	x		
Lack of control over power technology development, data security, legal control function	H2.3: The higher the share of commercial businesses in the MaaS scheme, the lower is political control of the applied technologies	x	x	x
High investment industry, high level of fixed capital, monopolization risks, fraudulent competition	H2.4: The higher the level of MaaS diffusion, the higher is the risk of monopolization			x
High investment industry, high level of fixed capital, low level of individual brand recognition, change of internal processes	H2.5: The higher the level of MaaS diffusion, the higher is the failure rate of individual businesses			x
Marginalization of lower-income, senior and fringe communities; technology gap; lack of political control	H2.6: The higher the share of commercial businesses in the MaaS scheme, the higher is the risk of marginalization of certain communities		x	

closer analysis reveals that many questions regarding its implementation, possible business models and actors involved remain unresolved.

This is also true with regards to the sustainability impacts of a further diffusion of the MaaS concept. Its potential contribution to a more sustainable mobility model is one of the main arguments for supporting this concept mentioned in both academic literature and public media. However, when collating the various statements citing possible effects to form hypotheses it becomes clear that the actual sustainability outcomes are highly uncertain. For example, the impact of MaaS on total greenhouse gas emissions within a specific region must be viewed as highly debatable, considering that different effects (e.g. reduction of congestion vs. shift away from public transport) may push the overall emissions in divergent directions. Similarly, new business opportunities, high levels of collaboration and the streamlining of operations are cited as major economic advantages of MaaS that may allow various businesses to operate more profitably. At the same time, however, the high level of investment required in the transportation industry, the large proportion of capital fixed in assets as well as low recognition of individual brands may lead to opportunities for monopolization as only existing larger players in the transportation industry are able to overcome these entry barriers.

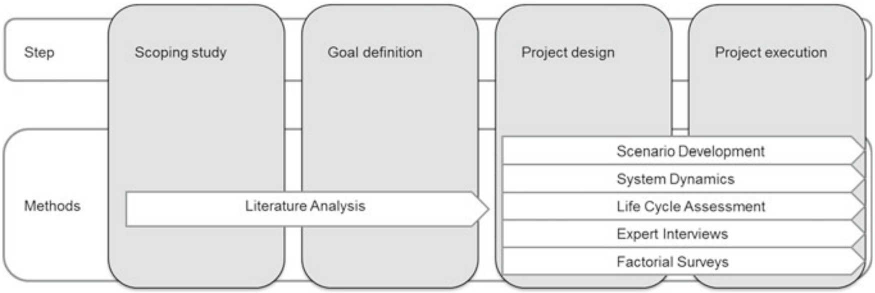

Fig. 3 Design of further TA project

These uncertainties confirm the necessity and relevance of a further in-depth technology assessment. In order to achieve greater clarity on the sustainability impacts of mobility as a service, the technology assessment needs to integrate a number of both qualitative and quantitative methods. In particular, the project design is to address the following research goals:

- Can the directions of the effects implied in the hypotheses be confirmed?
- How are the individual outcomes as stated in the literature interrelated? What are causal links between individual outcomes?
- How can the magnitude of the effects be estimated accuretaly?
- What are potential rebound effects? How can their magnitude be estimated accurately?

In order to adequately address these research goals, we propose the project design presented in Fig. 3. This design should be understood as an initial conceptualization, listing possible methods suitable for addressing the objectives defined for the respective impact categories. As noted previously, the main purpose of technology assessment is to evaluate a technology's impacts based on all information available at the time. Its effectiveness therefore depends on the constant interaction with and involvement in the respective topic. New developments, technological progress, related scientific findings as well as business cases must therefore be acknowledged and may lead to changes in the project goals, specific focus areas or the methods employed.

Since it is unlikely that empirical evidence on the impacts of MaaS diffusion will become available in the near future, any evaluation of both the direction of effects as well as their magnitude will depend on the development of scenarios. These may take the form of market scenarios portraying transition pathways of MaaS diffusion as well as scenarios incorporating anticipated developments in terms of technological progress, business model options, regulations and other political instruments (cf. [36]). The interrelations between the different effects may be presented and analyzed using system dynamics or other dynamic modeling techniques, which also provide useful starting points for analyzing potential rebound effects (cf. [37]). With regards to the established hypotheses, life cycle assessment (cf. [38, 39]) may be used to quantify environmental impacts of MaaS diffusion based on the previously developed scenarios, while expert interviews with stakeholders in the transportation industry may be used to evaluate economic impacts. Factorial surveys, allowing the reflection of complex decision-making contexts, may be suitable for further evaluation of the social impacts of MaaS diffusion (cf. [40]).

5 Conclusion

With the aim of improving understanding of the Mobility as a Service concept as well as the risks and opportunities associated with this scheme, this scoping study reviewed 95 sources of academic literature, grey literature and public media dealing with MaaS. The analyzed literature was synthesized to deliver six key elements that characterize MaaS schemes: multimodality, user centricity, cooperation, customization, technology integration and use of a unified platform. In addition, eleven hypotheses were formed regarding the potential environmental, social and economic impact of MaaS diffusion, which require further, more in-depth analysis in order to reveal causal relationships and magnitudes of the described effects. Scenario development, system dynamics, life cycle assessment, expert interviews and factorial surveys constitute possible methods for evaluating the sustainability impacts of MaaS diffusion in more detail as part of a future technology assessment. Overall, the necessity and relevance of an in-depth technology assessment project was confirmed.

Limitations may be found in the fact that the risks and opportunities discussed in this paper are derived solely from literature sources. Future research and in particular extended technology assessment projects would benefit from involving a diverse range of stakeholders of Mobility as a Service schemes in the assessment process in order to gain a more holistic understanding of the problem context.

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Dynamisation of Life Cycle Assessment Through the Integration of Energy System Modelling to Assess Alternative Fuels

Simon Pichlmaier, Anika Regett and Stephan Kigle

Abstract As greenhouse gas (GHG) emissions need to be reduced in order to limit the effects of climate change, Life Cycle Assessment (LCA) provides an internationally recognized framework to evaluate the environmental impact of energy supply and application technologies. However, standard LCA approaches are unable to depict the high dynamics of the future energy system. High shares of renewable energies and more variable loads intensify these dynamics according to a wide range of energy system scenarios. Therefore, a dynamisation and modularisation of the classic LCA approach is proposed in order to easily integrate the simulated electricity generation from energy system models on an hourly basis as well as future energy technologies. A special focus is put on Power-to-X (PtX) technologies in the transport sector due to its potential in deep decarbonisation scenarios.

Keywords Energy system modelling · Life cycle assessment (LCA) · Dynamic LCA · Power-to-X · Alternative fuels

1 Motivation and Problem Scope

The pathway to reach a sustainable supply of energy in the future is challenging. Nevertheless, a reduction of greenhouse gas (GHG) emissions is a pressing issue that needs to be addressed as soon as possible in order to stay below the limit of 1.5 °C of global warming [1]. The current primary energy supply is mainly based on fossil fuels. In 2016 fossil fuels accounted for 81.1% of the world's primary energy supply [2]. This applies globally as well as in Germany where in 2017 80.3% of the primary energy consumption was based on fossil fuels (hard coal, lignite, gas, oil) [3]. In order to fulfill the political agenda for decarbonisation a lot of research is conducted to illuminate the different routes towards sustainable energy supply.

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One main approach to model future energy supply are energy scenarios that are integrated in energy system models [4, 5]. A methodical framework is needed to assess the sustainability of these pathways. Life Cycle Assessment (LCA) provides such a framework and evolved to an accepted tool with international standards and norms [6], to evaluate the life cycle impact of different energy technologies [7, 8] and systems [9–11]. This article contains a detailed discussion of the requirements and limitations of the current state-of-the art of LCA of energy scenarios. It also makes a proposal to extend the current methodology to a dynamic and modulised LCA using the example of Power-to-X (PtX).

1.1 Deep Decarbonisation Energy Scenarios and the Relevance of Power-to-X Technologies

Most energy scenarios that are based on the political will to cut GHG emissions yield results that point into two slightly different directions—an electrification of energy applications or an electrification in combination with the production of synthetic fuels respectively PtX fuels [12, 13]. The main difference in whether the first or second alternative dominates, depends upon the level of decarbonisation. Especially ambitious levels of decarbonisation of up to 95% GHG emission reduction in 2050 compared to 1990, as for example aspired by the German government [14], favor a development of the energy sector towards an integration of PtX technologies [13].

The term PtX is not yet used uniformly in the literature and often causes confusion due to the variety of synonyms. Synthetic fuels, green fuels, renewable fuels, alternative fuels or electrofuels are just some of the many terms referring to fuels provided by similar technologies. In the present case PtX describes any technology that produces liquid or gaseous fuels which are produced from electricity. The key technology is the electrolysis for the separation of water into hydrogen and oxygen [15], which can be subdivided into alkaline electrolysers (AEL), proton exchange membrane (PEM) electrolysers and solid oxide electrolysers (SOEC). They differ mainly with regard to their efficiency, charge carriers and operating temperatures [16].

The resulting hydrogen can then either be used directly as a final energy carrier [17] or, in a further step, be synthesised into a carbon-containing energy carrier [18] (see Fig. 1). Currently, in literature there are two main synthesis routes for liquid energy carriers, namely methanol and Fischer-Tropsch synthesis [19]. Methanol synthesis uses CO_2 in combination with hydrogen as process inputs [20]. In the case of Fischer-Tropsch synthesis, CO is required. Therefore, CO_2 has to be converted to CO in a preceding reverse water gas shift reaction [21]. For the production of gaseous carbon-containing fuels methanation is one possible synthesis process to convert hydrogen to methane [22].

In any case, a CO_2 source is necessary. For an ultimately CO_2-neutral energy carrier, the CO_2 must not be obtained from the combustion of fossil energy sources.

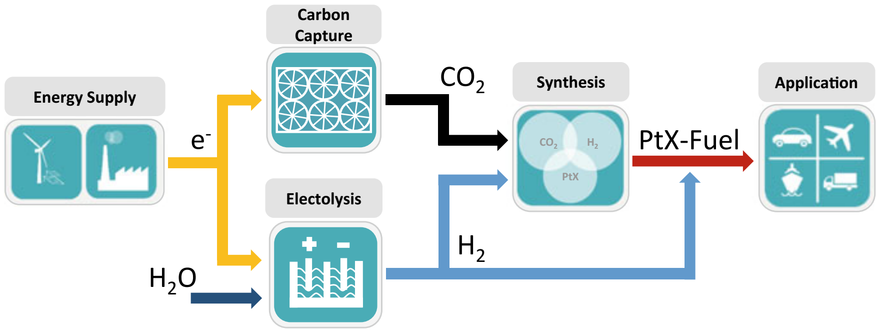

Fig. 1 Schematic representation of the production of PtX fuels

Direct air capture, the combustion of biomass or process emissions in the industry are suitable options [23].

For ambitious levels of decarbonisation these fuels are important as they possess two distinct advantages in contrast to the direct use of electrical energy:

1. PtX fuels have a high energy density. A high energy density is crucial in applications that are critical in respect to space or weight limitations such as e.g. the aviation sector.
2. PtX fuels offer the opportunity to store electricity surpluses from renewables over long periods of time.

Even though the energy efficiency is significantly lower than in case of a direct use of electrical energy [20–22], the advantages are taken to be crucial for an ambitious decarbonisation scenario.

Due to the fact that these technologies have not yet been assigned such a major role in the energy system transformation process, sustainability analyses are very rare in the literature. In addition, PtX is integrated into a highly dynamic energy system. A static assessment assuming a constant power supply is only valid to a certain degree. In the case of larger PtX plants, the repercussions on the energy system would also have to be considered. Furthermore, even with the integration of small plants and the associated marginal change in load without an effect on the energy system, it is not possible to depict the effects using a classical LCA.

1.2 Limitations of Classic LCA Approaches and the Development Towards Dynamic LCA

LCA methodology is commonly used to quantify the environmental impact of products or service systems [24]. For liquid and gaseous energy carriers a lot of research has been done on biofuels [25]. PtX fuels, however, only recently gained more attention. Additionally, only few research has been done on dynamic LCA such as e. g. the integration of LCA methodology into time-dependent energy system modelling.

A big hurdle to apply LCA to complete energy systems is missing data on future key technologies and production plants [10]. Furthermore, there are methodical limitations to standard LCA approaches such as double counting and shifting system boundaries in the analysis of the total energy system [11]. Another problem of LCAs of energy systems is more inherent to the energy system itself. In the context of a decarbonisation of the energy system a greater amount of renewable energies will be integrated into the power plant park. This integration will lead to higher dynamics in the provision of electricity as electricity from wind and photovoltaic plants is subject to volatile generation. However, not only the total amount of generated electricity and its provision is important, but also the flexibility of the demand-side technology and its ramp-up time to react to changes in the load. This may especially be important for the production of PtX fuels as their economic profitability strongly depends on the full load hours [26]. Therefore, the often used assumption of just deploying excess electricity from renewables is not strictly valid, but may possess potential for PtX fuels [27]. For it to be specified, a time-resolved consideration of renewable electricity generation and resulting operation strategies for PtX plants are required.

In the following a modular, dynamised LCA approach is proposed aiming at providing a more detailed assessment of future energy scenarios and the problem of volatility. The developed methodology is applied to the described case of PtX technologies.

2 Developed Methodology and Exemplary Application

According to DIN 14040/14044 an LCA can be structured in four phases: the definition of goal and scope, the life cycle inventory (LCI), the life cycle impact assessment (LCIA) and the interpretation. In Sect. 2.2 a method is explained to modularization and dynamics the determination of the LCI in order to assess the production and application of a PtX fuel in transport. The modularization allows different system boundaries and therefore different scopes of the LCA. This topic is discussed in Sect. 2.1. As an exemplary application of the methodology, in Sect. 2.3, the production of a PtX fuel with a dynamic energy system providing the electrical energy input is discussed.

2.1 Discussion of Goal and Scope for the Proposed Methodology

The proposed methodology does not imply one particular goal. Moreover, the goal has to be defined as soon as the level of modularisation is set. If the environmental

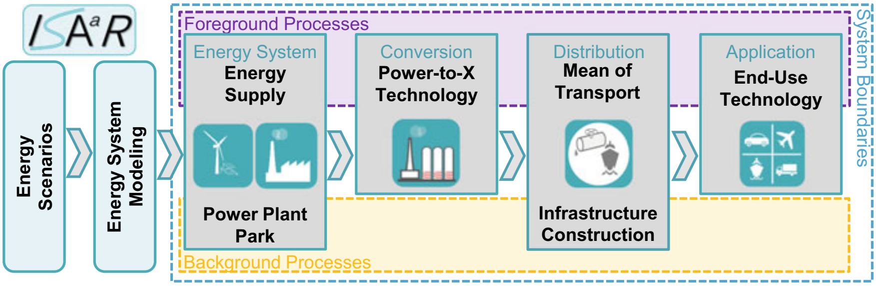

Fig. 2 Modularisation for the LCA of PtX fuels for transport applications with integration of the energy system model ISAaR

impact of the provision of 1 passenger-km (pkm) wants to be assessed, the scope and therefore the level of modularisation has to be set accordingly. But also parts of the life cycle, such as the provision of 1 kWh PtX fuel, can be assessed as explained in the following chapters. The scopes in these two cases are different and hence the system boundaries are different. Consequently, the modularisation adds flexibility to the choice of system boundaries.

In the following, first, Sect. 2.2 explains the method for the system boundaries according to a Well-to-Wheel analysis. Consequently, in addition to the operational considerations, the upstream chains of the individual life cycle stages are also considered. However, in Sect. 2.3 the focus is set on the fuel supply and the assessment method is shown for a functional unit of 1 kWh PtX fuel.

2.2 *Modularisation and Dynamisation of LCA*

Looking at the whole life cycle of a PtX fuel, it can be divided into energy supply, conversion, distribution and end use. Figure 2 illustrates these four steps within the system boundaries. It also shows which parts of the fuel life cycle are regarded as foreground and background processes. The foreground processes are part of detailed considerations, while the background processes are assessed using the database ecoinvent 3.5.

The energy system model ISAaR, developed at the Research Center for Energy Economics (FfE), provides the hourly German energy supply embedded in a European energy system. Among others, the energy sources considered include electrical energy, district heating and gas. By setting the boundary conditions, including the energy consumption by energy carriers, the dispatch of the supply systems for scenarios up to 2050 can be simulated [28] (see also Fig. 4). Additionally, the results contain economic as well as ecological indicators such as electricity prices and GHG emissions. The output of ISAaR in hourly resolution serves as an input for the LCA. Hence, the use of fuels for power supply can be considered a foreground process. This

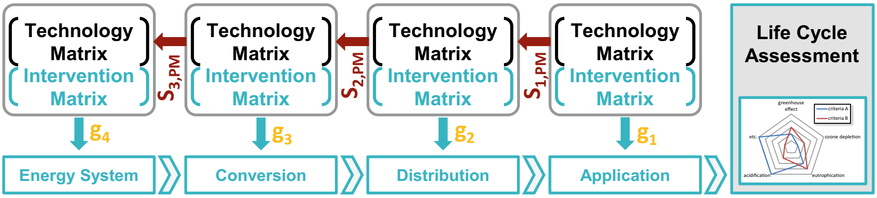

Fig. 3 Matrix interpretation of the modules to carry out the LCA. The part of the scaling vector described as $s_{i,PM}$ is the one to be handed over to the preceding system [see also Eqs. (4) and (5)]. The inventory vector g_i can be derived for each module

enables the time-dependent assessment of the environmental impact of the further PtX fuel production in the foreground and thus the inclusion of possible peaks in the power supply of volatile renewable energies. In addition, the applications such as cars and other means of transport are also considered in the foreground. This allows the comparison with transport applications using other energy sources with less favorable storage properties, such as electricity. In addition to the modeled foreground processes, the background processes are assessed with the help of the ecoinvent 3.5 database. Examples of background processes are the construction of required plants and means of transport as well as the provision of raw materials for power generation.

The modularisation of the LCA along life cycle phases is carried out to facilitate data integration and contribution analysis. It enables the smooth modification of known modules as well as the easy integration of new modules from e.g. PtX implementation projects. Furthermore, this approach allows individual sub-areas of the life cycle to be assessed in a transparent way, which simplifies the identification and communication of possible drivers of environmental impacts. As shown in Fig. 3 each module is represented by a separate technology matrix.

The matrix representation follows Heijungs et al. [29]. Therein the square technology matrix A is used to carry out the inventory analysis. A contains the input processes associated with a unit output process. For example, 1 m^3 hydrogen is assigned a certain amount of power generation. Given a final demand f the scaling vector s is to be carried out with Eq. (1):

$$s = A^{-1} f \tag{1}$$

The scaling vector serves to scale the unit processes of the technology matrix. With the use of s and the intervention matrix B, it is then possible to calculate the environmental flow vector g:

$$g = Bs \tag{2}$$

The emissions related to each unit output process are described in the intervention matrix B. It contains for example the CO_2 emissions of the combustion of coal to produce 1 kWh of electricity.

To connect the different modules, the elements in the scaling vector s of the processes to be considered in a preceding module are handed over as elements of a new final demand vector f. This concept is clarified in the following chapter using the example of a PtX fuel.

Eventually, all environmental flow vectors g_i are cumulated to derive the LCI. Ultimately, based on the LCI the LCIA can be conducted for each module separately or for all modules collectively.

2.3 Exemplary Application for the Production of PtX Fuel

The case of PtX fuel production is now considered more closely as an exemplary application of the methodological approach explained before. For this purpose, two systems of equations are set up. One represents the production of a PtX fuel and the other one describes the generation of electrical energy. The output of the whole system is a PtX fuel with the functional unit of 1 kWh referred to the lower heating value. According to Eq. (1) the first system of equations consequently results as follows:

$$\begin{pmatrix} a_{11,PtX} & a_{12,PtX} & \cdots & a_{1n,PtX} \\ a_{21,PtX} & a_{22,PtX} & \cdots & a_{2n,PtX} \\ \vdots & \vdots & \ddots & \vdots \\ a_{m1,PtX} & a_{m2,PtX} & \cdots & a_{mn,PtX} \end{pmatrix} \begin{pmatrix} s_{1,PtX} \\ s_{2,PtX} \\ \vdots \\ s_{n,PtX} \end{pmatrix} = \begin{pmatrix} f_{1,PtX} \\ f_{2,PtX} \\ \vdots \\ f_{n,PtX} \end{pmatrix} = \begin{pmatrix} 1\,\text{kWh} \\ 0 \\ \vdots \\ 0 \end{pmatrix} \tag{3}$$

Thereby the first row of A_{PtX}, s_{PtX} and f_{PtX} represent the PtX fuel production process. The final demand vector consists of the functional unit in the first row and zeros in every other row. Each column in the technology matrix A_{PtX} represents a unit output process. The rows contain the respective input processes. For the step of fuel conversion, the technology matrix contains, among others, the processes of operating materials such as water, CO_2 and electricity as well as the construction of the synthesis plant. Every entry a_{ij} of the matrix where $i = j$ is one. The system of equations is used to calculate the scaling vector as described above. The scaling vector contains entries of processes assessed in the PtX module and processes to be handed over to the previous module. The processes which are not handed over are used to determine the environmental flow of the PtX module g_{PtX}. For the other processes the resulting scaling vector elements are handed over as the final demand vector element of the preceding system of equations (see Fig. 3). Thus, if row α in Eq. (3) represents the provision of electrical energy, the scaling vector can be divided into one to be considered in the preceding module (PM) and one to be considered in the current module (CM):

$$s_{PtX} = s_{PtX,PM} + s_{PtX,CM} = \begin{pmatrix} 0 \\ \vdots \\ s_{\alpha,PtX,PM} = s_{\alpha,PtX} \\ \vdots \\ 0 \end{pmatrix} + \begin{pmatrix} s_{1,PtX,CM} \\ \vdots \\ s_{\alpha,PtX,CM} = 0 \\ \vdots \\ s_{n,PtX,CM} \end{pmatrix} \tag{4}$$

Consequently, the final demand vector of the energy system (ES) results in:

$$f_{ES} = \begin{pmatrix} f_{1,ES} = s_{\alpha,PtX} \\ 0 \\ \vdots \\ 0 \end{pmatrix} \tag{5}$$

Thus, the first row in the system of equations of the energy system corresponds to electrical energy. All other elements of the final demand vector are zero. With the use of the technology matrix of the energy system A_{ES}, the underlying environmental flow for the supply of electrical energy can now be determined as explained in Eqs. (1) and (2).

However, the high dynamics of the energy system must be included. Figure 4 illustrates the time dependency of the technology matrix of the energy system. A simulated dispatch for the provision of electrical energy for the year 2030 using the energy system model ISAaR is depicted.

The electrical energy generation of power plants is shown divided by type, such as i.e. gas, lignite and hard coal. In addition, the load and the export of electrical energy are presented. It can be seen that the deployment of the different types of power plants is subject to great variability. As an example, the technology matrices of two hours $t = 2300$ and $t = 2700$ are set up.

As a consequence, the resulting environmental flow will also vary over time. Therefore, the described system of equations must be solved for each time step, in this case each hour of the year.

Since in this example the energy system is the last set of processes to be considered, the cumulative environmental flow of the overall system for each time step can finally be calculated as follows:

$$\boldsymbol{g} = \boldsymbol{g}_{PtX} + \boldsymbol{g}_{ES} = B_{PtX} \cdot s_{PtX,CM} + B_{ES} \cdot A_{ES}^{-1} \cdot f_{ES} \tag{6}$$

As the formation of the inverse of a large, sparsely occupied matrix is a complex arithmetic operation, it is important to keep the technology matrix A_{ES} and therefore also the intervention matrix B_{ES} and the final demand vector f_{ES} as small as possible. In contrast to the consideration of the whole system, the modularisation and the associated partitioning of the technology matrix already allows a significant reduction of complexity. Additionally, the second term can be cleared of unused processes.

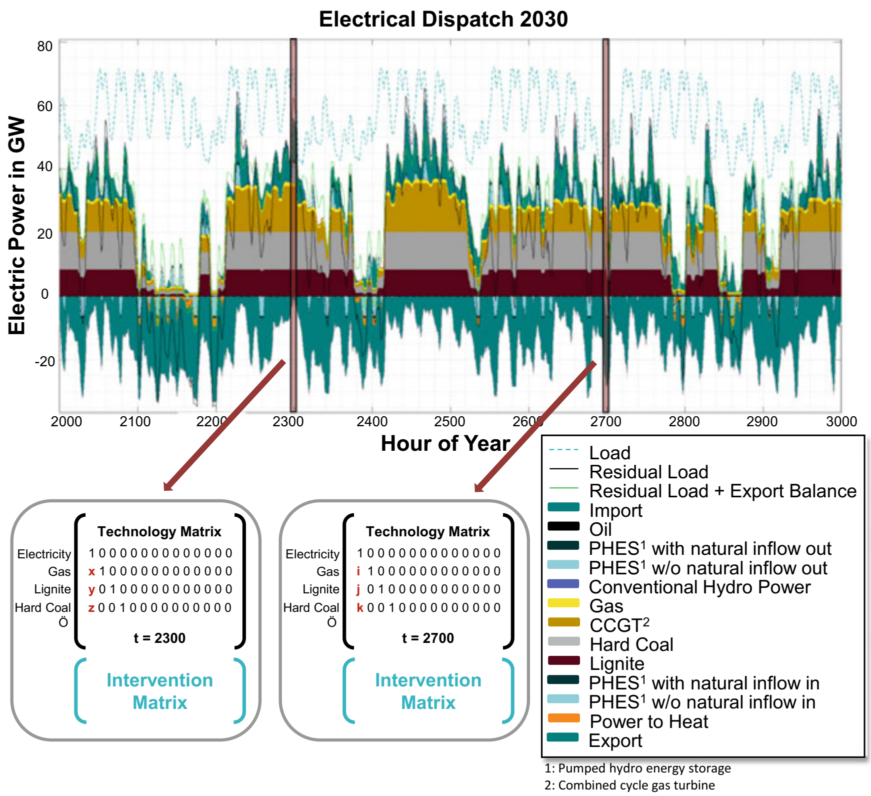

Fig. 4 Exemplary electrical dispatch for the hours 2000–3000 of the year 2030 and schematic representation of the technology matrices for two hours (t = 2300, t = 2700)

Then, in order to enable the addition with the vector g_{PtX}, the vector g_{ES} must again be extended by the corresponding zeros to match the length of the vector g_{PtX}.

In the present case, it is assumed that only the energy system varies over time. For example, in the event of a variable operation of an electrolyser, the system of equations for the conversion can also be time-dependent. In this case, the existing system can also be extended by a time dependency of the load. Therefore, e.g. partial load capable electrolysers can also be investigated.

3 Outlook and Future Projects

The methodology proposed above adds new aspects to classic LCA approaches. By being able to integrate distinct time series for the production of electricity and the related energy generation parks, a new level of dynamisation can be reached. The high temporal resolution adds to the accuracy of the LCA as the highly fluctuating composition of the electricity mix from different generation technologies can be taken into account. Additionally, the environmental impact of different electricity generation technologies can be compared according to different future energy scenarios.

The modularisation of the LCA adds the advantage to easily interchange, modify and add process data without the need to calculate the whole fuel life cycle all over again. Therefore, competing life cycles can easily be compared with regard to their environmental impact and new data of future technologies can be integrated to track environmental improvements. In addition, the computational complexity for the formation of matrix inversions can be reduced by the modularisation.

This newly developed methodology is implemented, in a first step, in a case study of different PtX production chains in the context of the project BEniVer. In a second step, a full integration into the energy system model ISaAR is aspired. By feeding the results of the LCA back into ISaAR it is possible to use them as an optimisation criteria in the plant expansion and deployment planning.

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Part III
Energy

External Costs as Indicator for the Environmental Performance of Power Systems

Lukas Lazar and Ingela Tietze

Abstract Power system planning progressively demands integrated assessment methodologies to meet the requirements of environmental sustainability goals. An approach to include environmental impacts into power system decision procedures is the use of external costs. To investigate the applicability of external costs for the environmental assessment of power systems, we integrate external costs into the method of Life Cycle Assessment (LCA) on the case of power generation technologies. The correlation between the LCA results considering external costs on the one hand and on the other hand standard midpoint impact assessment is investigated by regression analysis. We found that eutrophication (marine and terrestrial), acidification, photochemical ozone creation, respiratory effects and climate change show correlation ($R^2 = 0.97$–0.66). In contrast, the categories concerning land and resource use are not correlating. The correlation mainly depends on the elementary flows which are accounted for. External costs lack in including the variety of elementary flows which are considered in the midpoint assessment. An application of external costs as sole impact indicator of power systems is not recommendable at the current state of development and further research activity for the use in LCA is proposed.

Keywords External costs · Life Cycle Assessment · Impact assessment · Power systems · Sustainable energy

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1 Introduction

Human development has been accompanied by an extensive use of energy resulting in human activity's dependency on highly-concentrated fossil sources [1]. Today energy is non-substitutable for any modern economic activity and accordingly considered as additional factor of production [2]. By average one person consumed around 21 MWh of energy in 2017 with a range starting from 4 MWh in African countries rising to 90 MWh on the Northern American continent [1, 3]. This spread not only shows the distributional injustice of the use of conventional energy resources but also the prospective energy demand growth in the future, momentarily linked with the potentially rising environmental burdens. Energy generation by fossil fuels contributes significantly i.a. to global warming, acidification and particulate matter emissions. Between 2000 and 2010 the energy sector emitted 47% of the worldwide anthropogenic greenhouse gas (GHG) emissions [4]. In Germany more than 70% of the GHG emissions [5], 67–75% of the acid-forming NO_x- and SO_2-emissions and more than 40% of particulate matter emissions are assigned to the energy sector [6]. Already today human activity's effect on the ecosystem is measurable, pushing the earth outside the state of Holocene's stability [7, 8]. Hence, decision-making in the energy sector—so far mainly driven by economic factors—more and more calls for the implementation of environmental performance indicators into the assessment methodology [9–11].

For this purpose, Life Cycle Assessment (LCA) has been extensively applied to analyse the environmental burdens of energy technologies and systems. Due to the manifold dimensions of environmental burdens, the results are often provided as an array impeding the direct implementation into decision making [12, 13]. An approach, to overcome these obstacles, is the expression of environmental burdens in terms of external costs. External costs occur when the social or economic activities of one group causes a loss in welfare to another one and does not (fully) compensate this change [14, 15].

Even though external costs have been applied in several projects [14–18], their reliability compared to the LCA approach has not been proven yet. If external costs correlate to midpoint indicators used in LCA, their application to the environmental assessment can be of high interest. For instance, in the field of energy system modelling, costs are the determining factor for the optimisation. The implementation of external costs enables the direct inclusion of the environmental dimension into energy system modelling. Thus, the main targets of this chapter are: to integrate the external cost calculation into the Life Cycle Impact Assessment of a case study (T1), and to analyse its correlation to the commonly accepted midpoint methods (T2). In order to achieve the targets, the chapter is outlined as follows: after an overview of the background in Sect. 2, Sect. 3 describes extensively the methodology applied to analyse the benefits and limitations of the external cost approach in LCA. The results for the German electricity system technologies are given in Sect. 4. After the discussion in Sect. 5, conclusions are drawn and an outlook on further research activities is given.

2 Background

Among environmental assessment methodologies LCA has been established as a systematic method to identify the environmental impacts of products or services. LCA is increasingly implemented into decision making methods [19–24] and as a core element for environmental policies [25, 26]. Within an LCA the whole life cycle is analysed from extraction of the raw materials to processing, usage, recycling and disposal of the materials. The international norms ISO 14040 and ISO 14044 standardise LCAs initiating a comparable and reproducible procedure [27, 28]. LCA arranges its input and output (elementary) flows in the so called Life Cycle Inventory. These flows are aggregated and characterized following the Life Cycle Impact Assessment to estimate the final effect of each flow according to the respective impact category representing the environmental intervention. Impact categories are differentiated by the point of assessment on the underlying impact pathway. Midpoint impact categories such as CML [29], ReCiPe[1] [30], ILCD [31] are located at an intermediate point on the impact pathway, providing a higher certainty but a lower force of statement compared to endpoint impact categories. The point of impact of endpoint categories lies on the final effect or damage. Hence, in the environmental assessment of energy systems, Life Cycle Impact Assessment at midpoint level delivers a comprehensive but partly ambiguous result array. Weighting and aggregation procedures are not compliant to DIN ISO standards in LCA, therefore the communication of the variety of midpoint results remains difficult. Endpoint methods such as Eco-indicator [32], LIME[2] [33] and EPS [34] can provide a single score requiring normalisation and weighting mechanisms which can distort the initial results of the assessment.

External cost calculation can be considered as endpoint indicator delivering a single score in monetary terms which inheres the advantage of a possible linkage of environmental targets and economic factors. Moreover, from an economist's point of perspective, external costs arise out of the concern that environmental impacts of human activities, like energy use, are not being properly integrated into decision making and not reflected in the market price even though they can be irreversible. This market imperfection shall be resolved by the internalisation of externalities into the market mechanisms. Therefore a valuation of resources, goods and emissions which are withdrawn from or released into the ecosphere is implemented with the expectation to achieve a balancing between investment, operational costs and the costs of the environmental damages [15].

For power generation technologies the European Commission initiated several projects to ease decision-making by the implementation of external costs into the assessment. The ExternE study quantifies external costs by analysing nuclear, fossil and renewable fuel cycles for the externalities associated with electricity generation [14]. Impact effects of air pollution on the natural and human environment, consequences of accidents in the workplace, impacts of noise, visual intrusion and

[1]ReCiPe is also available as endpoint method (derived from the midpoint assessment) but is commonly used on the midpoint level.

[2]LIME includes midpoint categories which are derived from the endpoint methodology.

the effect of climate change were considered. ExternE aimed to find the optimal level of emissions by integrating the external costs into policy-making procedures, e.g. via taxation. For the calculation of site-specific damages, ExternE applies the impact pathway approach which considers the creation of secondary pollutants such as sulphates, nitrates and ozone depending on meteorological conditions, population distribution and also on the background concentration of the reactants such as NO_X, SO_2, NH_3, NMVOC. Therefore the impact pathway starts with the emission of a pollutant at the location of the source into the environment, it models the dispersion and calculates the impact to the receptor. The ExternE methodology is widely accepted nonetheless uncertainties and omissions of impacts asked for further research [15].

NewExt [18] focused on improving ExternE's methodology in terms of monetary valuation, impact valuation and weighting, multi-media impacts and quantification of major accidents in non-nuclear fuel chains. ExternE-Pol [17] updated, improved and extended ExternE's methodology and verified it by the application on the power production in the Czech Republic, Hungary and Poland. The NEEDS project firstly integrated private and external costs within one dynamic framework [16]. It had the objective to evaluate the full costs and benefits of energy policies and future energy systems, both for individual countries and for the enlarged European Union as a whole. NEEDS aimed at performing LCAs of new energy technologies, to develop and improve monetary valuation of externalities associated to energy production, to integrate LCA and external information into policy formulation and to examine the robustness of the proposed technological solutions in view of stakeholder preferences. The focus intended to provide direct, usable inputs to the formulation and evaluation of energy policies accounting the economic, environmental and social dimension of sustainability. Within NEEDS the project CASES analogously investigated external and internal costs of electricity generation for different energy sources at national level for the EU27 countries also in a future perspective of 2030 [15]. CASES evaluated policy options for improving the efficiency of energy costs by taking into account the full cost data comprising investment, operational and external costs. Building on the results of ExternE, NEEDS and CASES, the web tool EcoSenseWeb[3] provides a calculation of external costs related to the exposure to airborne pollutants with the goal to provide information about air pollution mitigation strategies [35].

3 Methodology

According to the targets of the study, the methodology can be further subdivided into three parts as illustrated in Fig. 1: Adaption and linkage of external costs into the LCA software and database (T1a), conducting LCAs for a case study of power gener-

[3]Ecosense web tool can be assessed online http://ecoweb.ier.uni-stuttgart.de/EcoSenseLE/current/index.php.

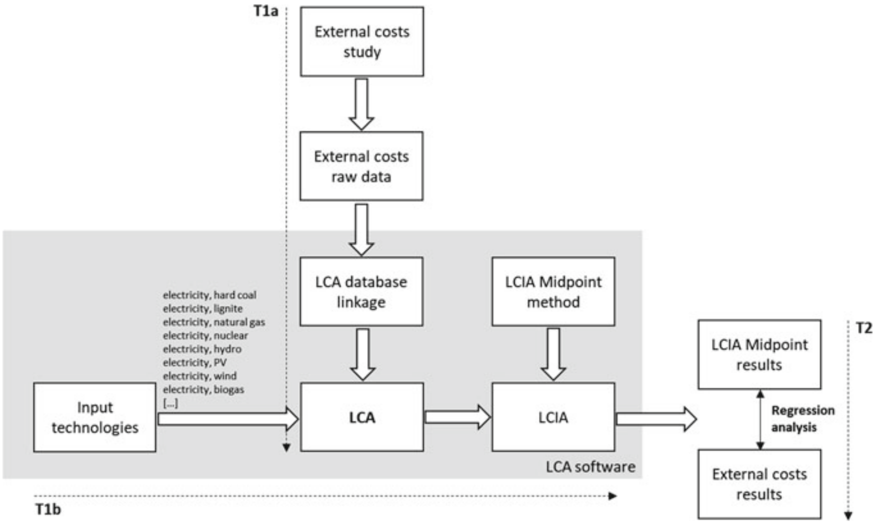

Fig. 1 Methodological framework including the necessary steps to fulfil the targets T1a, T1b and T2. The steps in the grey box illustrate the steps within the LCA software

ation technologies applying midpoint indicators and external costs (T1b), correlation analysis between midpoint indicators and the external costs (T2).

3.1 External Cost LCA Database Integration

From the external cost approaches, the CASES study was used for further considerations due to transparency reasons and data availability [15]. The results of the external cost calculation are integrated as Life Cycle Impact factors to corresponding ecoinvent materials. The subdivision followed the CASES methodology separating emissions in three heights of release levels (high >100 m, low <100 m, unknown/unspecified). Elementary flows in ecoinvent allocated to "lower stratosphere + upper troposphere" are excluded because they were not covered in the CASES study. Radionuclides were subdivided by the medium in which they are released (air/water) according to the CASES dataset. Rubidium-106, originally integrated in CASES, was excluded as no ecoinvent correspondent was available.

3.2 LCAs of Power Production Technologies in Germany

The results of the LCAs are used to compare midpoint impact category indicators with the external cost approach developed in the CASES project [15]. The study follows to a great extent the ISO standards 14040, 14044 and ILCD recommendations [27, 28, 31, 36, 37]. The LCA software Umberto LCA+ along with ecoinvent's cut-off system model are used for modelling, according to an attributional approach of the study not considering any benefit for recycling processes.

The LCAs are carried out to assess the environmental performance of power generation technologies implemented in ecoinvent's electricity grid mix for Germany. The German power system is used as a first example for an energy system comprising a mix of fossil, nuclear and renewable power generation technologies.

The main function of the different power generation technologies (product systems) is the provision of electricity ready for the grid, hence high voltage electricity in cradle-to-gate perspective is assessed. Data which is included in the process "market for electricity, high voltage [DE]" was selected. The product systems exclusively consist of ecoinvent 3.4 data (see Electronic Supplementary Material) and is representative for the year 2014. Technical representativeness including the correspondent plant life time depends on the individual ecoinvent dataset. Recycling was not added due to unknown future recycling procedures.

The functional unit of all product systems is the "production of 1 kWh electricity, high voltage". Analogously the reference flow requires an individual amount of the corresponding energy source (e.g. fuel, uranium, wind, solar irradiation) and a technical system to provide the functional unit. Input data of the LCA is solely based on ecoinvent 3.4, hence data quality is constituted by the data quality guideline [38]. Recent ILCD recommendations for LCA in the European context are used for the midpoint impact assessment [31]. For the external costs an impact assessment based on data of the CASES study is implemented in the LCA software connected with the ecoinvent database. The method is named XTCosts and uses €$_{2000}$ as unit of measurement (according to the external cost assessment of the CASES study valid for the year 2000). The calculation follows Life Cycle Impact Assessment methodology, hence environmental impacts are calculated by:

$$\mathrm{EI}_{\mathrm{p,c}} = \sum_{i=1}^{\mathrm{n}} \mathrm{e}_i \cdot \mathrm{q}_i \tag{1}$$

where $\mathrm{EI}_{\mathrm{p,c}}$ is the environmental impact of product system p for the impact category c (e.g. climate change, etc.), e_i the environmental intervention (e.g. emission or extraction of resource) and q_i the corresponding characterisation factor, dependent on the intervention. The external costs $\mathrm{XTC}_{\mathrm{p}}$ are calculated analogously, ending in a single indicator not demanding any category subdivision:

$$\mathrm{XTC}_{\mathrm{p}} = \sum_{i=1}^{\mathrm{n}} \mathrm{e}_i \cdot \mathrm{p}_i \tag{2}$$

The characterization factor is substituted by the cost factor p_i in this equation. Similar to the characterization factor, this cost factor depends on the elementary flow and e.g. for emissions also on the height and medium of release.

3.3 Linear Regression Analysis

Linear regression analysis is applied using log-transformed data to compare the midpoint indicators with the respective external costs. Curve fitting showed that a power function increases the statistical coefficient of determination for some impact categories,[4] therefore a nonlinear relationship is chosen as basis for the analysis. The power function equals[5]:

$$y = \beta x^{\alpha} \tag{3}$$

with $y = EI_{p,c}$ and $x = XTC_p$.

To find the intercept β and slope α linear least-squares fitting was used. The coefficient of determination R^2 reveals the correlation of the datasets. Linear regression plots with 95% confidence are provided. The significance was tested by the strength of evidence against null hypothesis represented in the p value according to Fisher [39]. The coefficient of determination is defined by:

$$R^2 = 1 - \frac{\sum_{i=1}^{n}(y_i - \hat{y})^2}{\sum_{i=1}^{n}(y_i - \bar{y})^2} \tag{4}$$

$$\bar{y} = \frac{1}{n}\sum_{i=1}^{n} y_i \tag{5}$$

A data adjustment was applied to the ecoinvent datasets because of high ammonia outputs in the inventory. The ecoinvent process "heat and power co-generation, biogas, gas engine [DE]" has a high share of ammonia occurring in the anaerobic digestion process for the biogas production leading to skewed results. It is exchanged by the ecoinvent process "biogas production from grass [CH]".

4 Results

Concerning land and resource use, the calculations for photovoltaics show the highest impacts. Fossil power producers (coal, lignite, oil) have partly relative high outputs in the impact categories climate change, acidification, eutrophication and photochemical ozone creation. These results correspond to LCA studies of power production [12, 40]. External costs are high for fossil, biogas, wood chips and photovoltaic power generators and low for nuclear, wind and hydro technologies.

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[4] Acidification, ecosystem human and ecosystem ionizing radiation, marine and terrestrial eutrophication, ozone depletion potential, photochemical ozone creation, respiratory effects.

[5] For land use negative impacts occur and therefore log-transformation is not used. The result is validated with log-transformed data excluding negative values.

Table 1 Coefficient of determination R^2 between common midpoint impact categories and external costs (XTCosts), based on 22 power generation processes

Impact category	R^2	Impact category	R^2
Eutrophication, marine	0.9664	Human toxicity, carcinogenic	0.3584
Acidification, freshwater, terrestrial	0.9158	Ionising radiation, human	0.2350
Eutrophication, terrestrial	0.8798	Ecotoxicity, freshwater	0.2313
Photochemical ozone creation	0.8469	Ionising radiation, ecosystem	0.2225
Respiratory effects, inorganics	0.6937	Ozone layer depletion	0.1892
Climate change	0.6576	Resource use	0.0166
Eutrophication, freshwater	0.5497	Land use	0.0000
Human toxicity, non-carcinogenic	0.4577		

Table 1 shows the regression results for 15 midpoint impact categories and Fig. 2 an excerpt of six impact categories in relation to the external costs. The data points in Fig. 2 represent the 22 electricity generation technologies[6] from ecoinvent. The impact categories covering eutrophication, acidification, photochemical ozone creation, respiratory effects and climate change show a correlation with $R^2 = 0.97–0.66$ and *p* values less than 0.01%. The categories for freshwater eutrophication, human toxicity, ionising radiation, freshwater ecotoxicity and ozone depletion show low correlation ($R^2 = 0.55–0.19$) with a p value of less than 2.7%. The categories for land and resources use show no correlation ($R^2 < 0.02$).

5 Discussion

This investigation of the applicability of external costs as an indicator for the environmental performance of power systems is subject to limitations given by the methodology used and the assumptions made. The LCA methodology includes uncertainties in the data sources, dataset choices, specificity of the datasets, assumptions made, and methodological choices. Thus, the study at hand relies on the data quality as well as on the inherent allocation procedures of the database [38]. Moreover linear scaling of datasets and impacts do not have to correspond with the real-world behaviour which however would go beyond the methodological limits of LCA [41]. The topi-

[6]Full Life Cycle Impact Assessment results for the 22 electricity generation technologies can be found in the Electronic Supplementary Material including the ILCD midpoint impact categories and the external cost implementation (XTCosts).

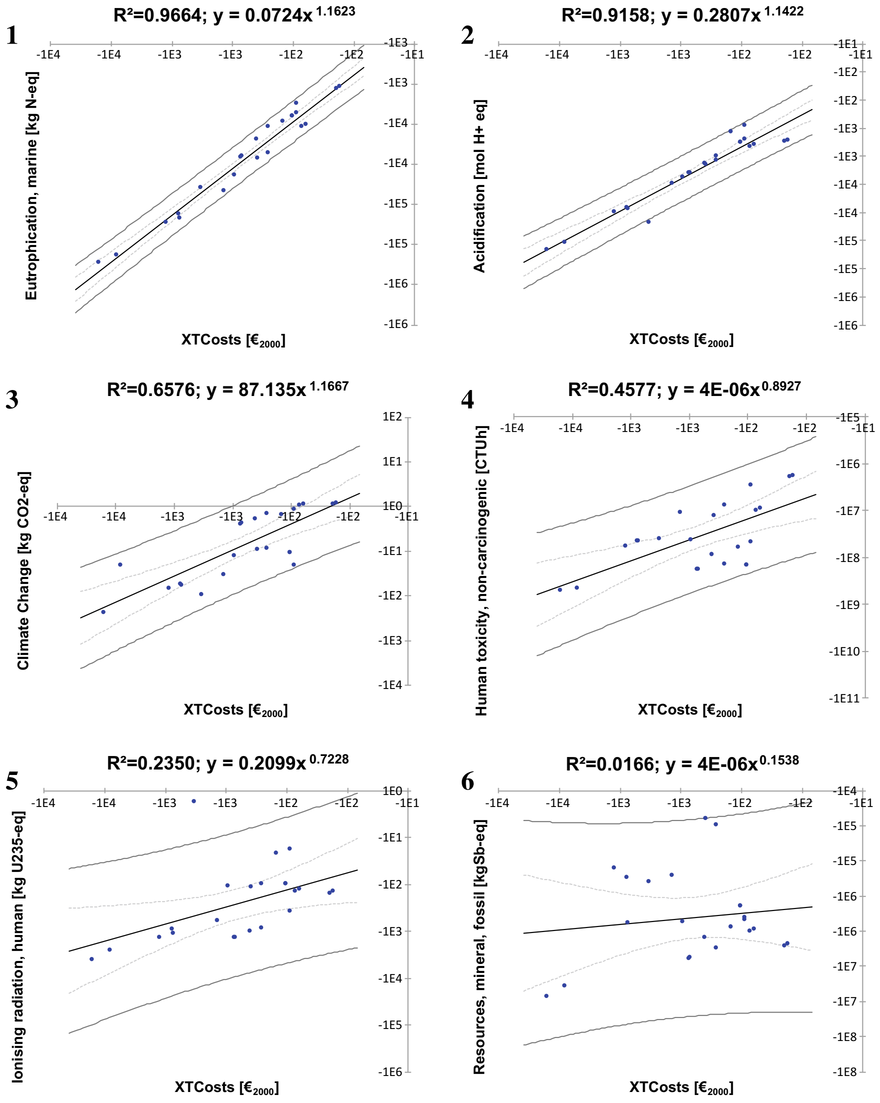

Fig. 2 Examples of linear regression lines for external costs (XTCosts) and eutrophication marine (1), acidification (2), climate change (3), human toxicity non-carcinogenic (4), ionising radiation human (5), resources mineral fossil (6), based on 22 power generation processes including 95% confidence (dotted lines) and prediction intervals (thin outer lines)

cality can be crucial especially on fast-changing technologies such as photovoltaics (which are based on the year 2012 in ecoinvent 3.4). Furthermore Life Cycle Impact Assessment methods mostly do not distinguish between local or global environmental interventions. However, in the assessment of energy systems local environmental burdens can have a fundamental impact.

The external cost assessment based on the CASES study additionally includes assumptions and data uncertainties which can influence the results. Exchanging the study for a different external cost approach could shift the results. The scope of the CASES project takes the national background pressure into account, by considering the country of emission. For emissions outside the national scope, the background pressure of the individual countries is not considered in the study at hand as ecoinvent does not provide geographic allocation of the environmental interventions. Thus, especially for renewable power generation technologies, the result can be misleading, because construction and material extraction plays a greater role than the environmental interventions during the operating time. The external cost assessment of the CASES study is based on models which do not take into account the full spectrum of environmental interventions represented by ecoinvent. The modelling at the endpoint perspective is far more complex and uncertain considering distribution, interdependencies and secondary processes. Moreover the CASES methodology states that impacts due to final deposition of radioactive waste were not evaluated. In contrast to that the ecoinvent database does cover radioactive waste treatment [42]. Current estimates anticipate additional costs for the final storage of radioactive waste [43]. Land and resource use were not represented directly in CASES, solely indirectly through ecosystem damage potentials of associated elementary flows.

The Life Cycle Impact Assessment results for the case study of power producers in Germany reveal that correlation between external costs and midpoint methods increases by the amount of the same elementary flows considered. E.g. in acidificiation ($R^2 = 0.918$) all contributing substances are taken into account both in the external costs assessment and in the midpoint categories (comprising nitrogen oxides, sulfur dioxide and sulfur trioxide). Impact categories showing a low correlation either account for elementary flows which are not considered by the external cost assessment or consist of multiple substances: For example human toxicity carcinogenic/non-carcinogenic consider more than 65/213 elementary flows in the midpoint assessment while only nine elementary flows are covered by the external costs. The impact category human toxicity carcinogenic is mainly dominated by the elementary flow chromium VI released into groundwater, which has been found specific for the underlying methodology and differs compared to other eco-toxicological impact assessment methods [44]. These discrepancies also appear in the assessment with zinc and arsen both released into groundwater being the main drivers for human toxicity non-carcinogenic. Uncertainties and an overestimation of zinc toxicity in the applied impact method is discussed by recent research. According to Nordborg et al. [45] uncertainties are large and the zinc overestimation remains a paradox that needs to be resolved. Further, zinc is not considered by the external costs, which could explain the non-correlating behaviour.

As an exception the midpoint impact categories human and ecosystem ionising radiation have similar elementary flows in both the midpoint and the external cost assessment but their correlation is low ($R^2 < 0.235$). In our analysis ionising radiation is the impact category with the biggest individual distance among the product systems steming mostly from the nuclear power generators. Exclusion of the nuclear power generator in our study increased the coefficient of determination to $R^2 = 0.44$. As the external cost assessment also includes all other substances not related to ionising radiation, we assume that substances other than radionuclides distort the results compared to the relatively small changes that occur between the non-nuclear power generators. Hence, if we separate the external cost assessment only accounting for radionuclides, a very high correlation between the ionising radiation categories and the external costs can be reached ($R^2 = 0.94$–0.97; $p < 0.01\%$). Ionising radiation is therefore reflected in the external costs with a relative low weight compared to other impacts.

For the remaining low or non-correlated impact categories (eutrophication freshwater, human toxicity cancerogenic and non-cancerogenic, ozone layer depletion, ecotoxicity freshwater, resource use and land use) the external costs—compared to the midpoint assessment—consider less elementary flows. However, impact categories such as resource and land use are difficult to convert into monetary values because the desired endpoint effect is unresolved. Moreover, especially renewable energies have lower impacts in categories such as climate change but at the same time increase e.g. in land use. This offset, occurring in the single external cost indicator, can potentially contradict the correlation. Additionally it has to be considered that regression analysis can be very sensitive with the limited dataset used (22 product systems) because outliers can influence the results more easily than it would be the case with a wider dataset [46].

6 Conclusions and Outlook

Under the assumptions made and the limitations given we conclude that the application of external costs as single environmental performance indicator is restricted and not recommended to substitute commonly used midpoint impact indicators. In the case of power generation technologies from the German electricity grid mix, external costs show potential applicability as a proxy for the impact categories marine eutrophication, acidification, terrestrial eutrophication, photochemical ozone creation, respiratory effects and climate change (in an descending order). For freshwater eutrophication, carcinogenic and non-carcinogenic human toxicity, freshwater ecotoxicity and ozone layer depletion potential the external cost assessment has to be further developed to achieve a higher correlation. Ionising radiation both ecosystem and human momentarily have to be presented separately as the inclusion is distorted by other elementary flows in the external cost approach. Furthermore, land and resource use are not included directly in the external cost assessment and it has to be stated that an inclusion remains challenging.

The data used for power generation technologies shows inconsistencies: a validation with further databases as well as the provision of a specific regional energy generation database would improve the power of interpretation. The continuation of correlation analysis for further product systems and impact categories is seen as necessary as well as the development of methodologies allowing a simultaneous or hybrid use avoiding the omittance of risks. Therefore a hybrid could be thought of in form of an endpoint indicator connected with a risk factor showing the maximum deviation of the product system's impact compared to alternatives. This approach could indicate a risk which motivates the recipient of the study to look deeper into the midpoint assessments if demanded. An environmental assessment by an external cost indicator is momentarily not seen as reliable enough to show the environmental performance of power systems for recommendation and decision-making procedures. Nevertheless, it represents a powerful additional indicator which can be easily integrated into energy modelling, decision and policy-making and thus further research is considered necessary in this field.

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A Spatially Explicit Life Cycle Assessment Tool for Residential Buildings in Lower Saxony: Development and Sample Application

Ann-Kristin Mühlbach, Michael W. Strohbach and Thomas Wilken

Abstract The ambitious goal of the German federal government to achieve a "nearly climate-neutral" building stock by 2050 should be underpinned by detailed knowledge on the whole life cycle of the housing stock in Germany. Therefore, a life cycle assessment (LCA) tool is developed, that combines the embodied energy (energy used for production of a building) as well as the energy consumption of existing buildings. By combining LCA data with a customized extract from the 2011 census for parts for Lower Saxony, the tool allows for spatially explicit assessments on a square kilometer grid. The classification of buildings, using building type and construction year, offers the possibility to quickly evaluate the building stock without the need for detailed information. In the future, the tool will be expanded to enable comparing the impact of actions like renovation of existing buildings on the one hand, and demolition with new construction on the other. Thus, scenarios can be analyzed and priorities for interventions identified. Combined with other information in regional sustainability assessments, for example mobility analyses and environmental impacts of land consumption, the tool will allow exploring paths to greater sustainability for the built environment.

Keywords LCA · Sustainable housing stock · Spatial data · Embodied energy

1 Introduction

Buildings are a major factor when it comes to human material and energy use [1], making them a key factor for reaching sustainability goals. In Germany, for example,

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private households account for about one third of the total end energy consumption [2]. Making buildings more energy-efficient is therefore a common sense measure for increasing sustainability and a major policy goal. In Germany, the federal government has set the goals for 2050 to reach a "nearly climate-neutral" building stock [3].

Life cycle assessments have shown that the operational phase contributes more than 50% of the greenhouse gas emissions of buildings [4]. The production of a building, however, also has a significant impact, and the energy used for construction of a building (around 30–40%) is currently increasing, due to the use of new materials for energy saving approaches [5]. Therefore, when estimating the environmental impacts of buildings, the whole life cycle must be considered.

When other aspects such as mobility or land consumption are included in sustainability assessments, the location of a building becomes a key factor [6]. Simply speaking, an energy efficient building with residents that commute long distances by car, is less sustainable than one that is located close to a public transportation hub. Hence, increasing sustainability of buildings has a spatial aspect and must include other disciplines, such as transportation sciences, environmental sciences and planning. The METAPOLIS project, of which this study is part of, takes such a holistic perspective [7].

In this chapter, a spatially explicit Life Cycle Assessment (LCA) tool for residential buildings is developed by combining LCA data for buildings with spatial census data. The connecting element is a grid of one square kilometer, which divides cities and villages in square patterns. By using this method, the combination of spatial and non-spatial data is possible. In this chapter, first results for two grid cells in the study region are presented and some implications are discussed. This is followed by a discussion on how the tool can be improved in the future.

2 Method

This section explains the method, which is used to develop a spatially explicit life cycle assessment tool for residential buildings in Lower Saxony. Two basic parts, the life cycle data of building classifications and spatial data (census data), will be connected to evaluate building stock (Fig. 1). Before going into detail, a short description of the case study area is presented.

2.1 Case Study Area

The METAPOLIS project studies two parts of Lower Saxony, Germany, including urban and rural areas (Fig. 2A). The eastern part includes the cities of Braunschweig, Wolfsburg and Salzgitter, with 251,364, 124,045 and 101,079 inhabitants in 2015, respectively. The western part is located south of Bremen and includes Vechta with 31,558 inhabitants as its largest city [8]. The rationale behind the split study region

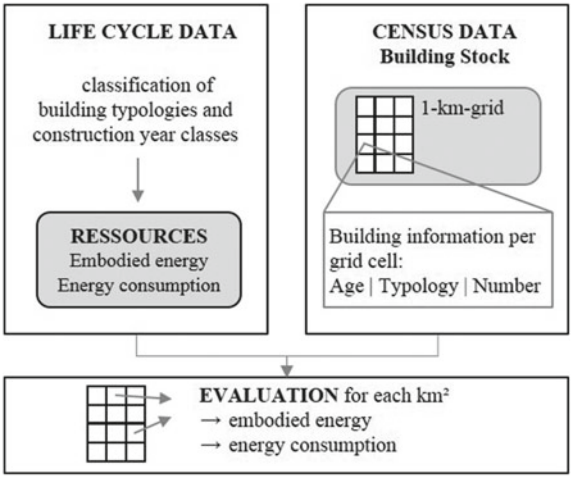

Fig. 1 Graphical representation of the spatially explicit life cycle assessment (LCA) tool which combines LCA data with spatially explicit census data

is a compromise between not being able to study all of Lower Saxony, while still covering two typical regions in respect to biogeography, industrialization, agriculture, and settlement structure. The total population of the region in 2015 was 1,671,926 and the total size is 911 km^2 [8]. It contains 437,113 buildings with residential use [9].

2.2 *Life Cycle Data*

Considering the large number of residential buildings in the study region, a classification had to be applied that allowed for the combination with LCA data. A classification of buildings (Table 1 and 2) has been used, based on a study by the *Institute for housing and environment (IWU)* [10], that takes building type and age into consideration:

For each combination of building type and year, a life cycle assessment was conducted with the *eLCA Tool* of the *Federal Institute for Research on Buildings, Urban Affairs and Spatial Development* [11]. The structure and building materials from IWU [10] were used to create a model of each building type in *eLCA*. The calculation was done with data from *Ökobaudat 2016,* a standardized database for ecological evaluations of buildings [12]. The functional unit is the net floor area [m^2] of each building. To evaluate the built environment, it is important to take the energy for the production of buildings into account. However, the eLCA Tool does not account for the production energy for existing buildings, only for new buildings. Therefore, the production energy for new buildings is added in the LCA of existing

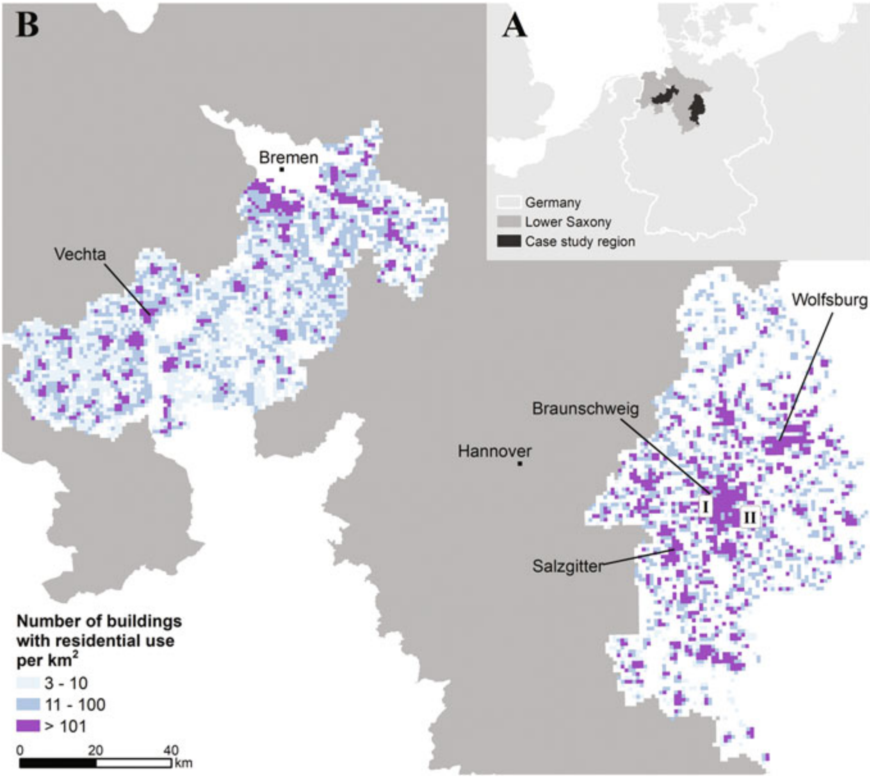

Fig. 2 The location of the case study area in Germany and Lower Saxony (**A**), and the number of buildings with residential use within a grid of 1 km^2 resolution (**B**). The roman numerals I and II near Braunschweig refer to the examples discussed in Sect. 3. Data A: ESRI base map Europe and BKG (2016). Data B: Census 2011 and BKG (2016)

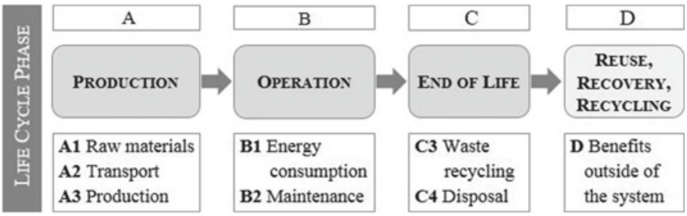

Fig. 3 The life cycle phases, based on [13]. So far, only phases A and B have been implemented in the LCA tool. Phases C and D will be implemented in the future

buildings. The life cycle phases (Fig. 3) were chosen following the structure of *German Sustainable Building Council* [13].

Life Cycle Phase "A—PRODUCTION" represents all environmental impacts for the production of building material, including raw material procurement, transporta-

Table 1 Building types used for the classification of the housing stock, based on [10]

Building types	
Single Family Houses (SFH)	Detached, 1–2 apartments
Terraced Houses (TH)	Semi-detached or terraced, 1–2 apartments
Multi Family Houses (MFH)	3–12 apartments
Apartment Blocks (AB)	13 or more apartments

Table 2 The construction year classes used for the classification of the housing stock, based on [10]

Construction year classes			
A	Before 1859	G	1979–1983
B	1860–1918	H	1984–1994
C	1919–1948	I	1995–2001
D	1949–1957	J	2002–2009
E	1958–1968	K	2010–2015
F	1969–1978	L	After 2016 (not used)

tion and processing. Only the shell construction (KG 300, DIN 276-1, [14]) is part of the calculation, since interior coating or furniture depends on the individual user and is not related to building types. The structure of interior walls is based on assumptions for standard bearing and non-bearing walls.

The phase “B—OPERATION” involves energy consumption by the user, as well as maintenance efforts. The electricity consumption by the user is estimated for an average three-person-household. The heat consumption depends on the building standard, which is why it was estimated based on average measurements [10]. The end energy use of each building type was multiplied with the primary energy factor for different energy sources. The allocation was done in accordance with information on energy sources for households in Germany [15] (Table 3). Furthermore, a renovation rate of 1% was assumed for existing buildings [16], while considering certain restrictions, for example façade insulation for historic landmarks, which could be protected for heritage purposes. For these buildings, the highest energy saving standard cannot be reached. A defined percentage for each construction year class will therefore be considered as not able to be renovated or only partially renovated, according to [3].

The phase “C—END OF LIFE” includes waste recycling and disposal of building material. Life cycle phase “D—REUSE, RECOVERY, RECYCLING” involves possible benefits outside of the system, such as reusing old materials for new products. Both parts (C and D) of the assessment will be taken into account for the development of future scenarios, for example demolition and new construction versus refurbishment. The evaluation of existing buildings, however, only takes production (A) and operation (B) into account.

Table 3 End energy sources for households in Germany, 2016, based on [15]

Energy sources			
Gas	40%	District heating	7%
Oil	19%	Renewable energy	13%
Electricity[a]	19%	Coal	1%

[a]Including renewable energies

2.3 Census Data

The census of 2011 is the most up-to-date census in Germany. It not only includes demographic data, but also information on buildings and dwellings, including building age, building type, size, number of rooms, heating type and vacancy rate. It is provided for general use on a municipal level and on a square kilometer grid (Fig. 1) [9]. However, in this form it is not possible to detangle different factors from each other. For example, the number of single houses and the number of houses built between 1919 and 1949 are provided, but not the number of single houses built in this time period. Therefore, customized data on the square kilometer grid level from the state office for statistics [LSN, pers. communication] was obtained, that could be linked to the classification of building types and construction year classes (see Tables 1 and 2).

The census follows strict rules in order to assure that direct or indirect identification of individuals or their personal or material situation is not possible. This is done by omission or a slight modification of data. For example, grid cells with very few buildings of a certain type and construction year class contain modified data or no data at all to ensure the privacy of individuals living in the region (see [17] for more information). Around 4% of all cells from the customized data have been omitted entirely for privacy protection reasons [LSN, pers. communication]. While the data has some limitations in areas with few houses and for combinations of housing type and construction year with few cases, major patterns of buildings are well represented at a high spatial resolution.

3 Sample Application

For demonstration, two example grid cells were selected, one in an urban area (I, Fig. 1B), which is dominated by multi-family houses (66%), and one in a suburban location (II, Fig. 1B) with mostly single family housing (75%). The grid cells contain 464 and 623 buildings, respectively [9]. The customized census data, however, only contains 430 and 500 buildings, respectively, due to privacy concerns [LSN, pers. communication]. Construction year classes around 1970 (~60%) dominate grid cell I, while grid cell II is mixed with ~40% from 1960 to 1970 and some new buildings built between 2002 and 2011. To generate a comparison of the grid cells on a per

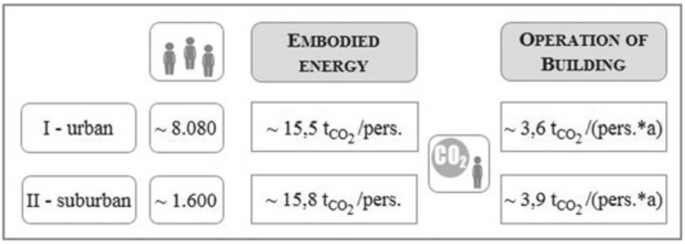

Fig. 4 The Global Warming Potential (GWP) per person (CO_2-eqv.) in the two example grid cells

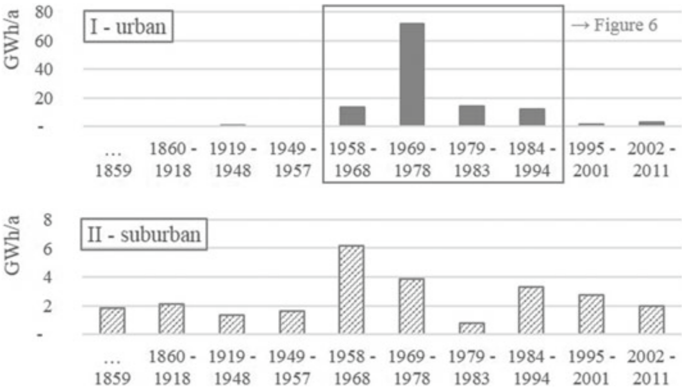

Fig. 5 Primary energy consumption per construction year class (operation phase) in the example grid cells I and II

capita basis, the human population had to be adjusted accordingly. A simple conversion factor has been chosen, based on the ratio of buildings in the customized census data and the number of buildings in the unadjusted census data. Thus, the 8.717 inhabitants of grid cell I were adjusted to 8.078 and the 1.989 inhabitants of grid cell II were adjusted to 1.596.

The CO_2 emissions or Global Warming Potential (GWP) per person during operation and the embodied energy per person are both slightly lower in grid cell I than in grid cell II (Fig. 4). Taking the consumption during operation in different construction year classes into consideration provides an overview on which construction year classes have the biggest impact (Fig. 5). The consumption of primary energy during the operation phase in grid cell I is less spread across construction year classes than in grid cell II, which makes it easier for a targeted intervention.

Going further into detail in grid cell I, shows that apartment blocks from the 1970s have the largest energy consumption and should be targeted for making the biggest environmental impact (Fig. 6) .The same type of analysis can be done for different parameter, as well as for other grid cells. The potentials of the tool will be discussed in the next section.

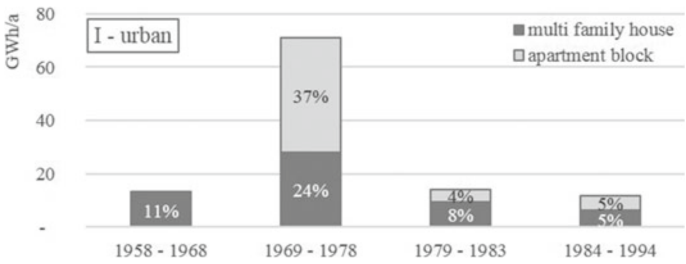

Fig. 6 Primary energy consumption per building type and construction year class (operation phase) in grid cell I

4 Discussion

The Life Cycle Assessment tool for residential buildings is a promising instrument for planning. It provides municipalities with an overview on the status-quo and a perspective on the most promising paths to reach a "nearly climate-neutral" building stock. Because it is spatially explicit, it allows for delineating redevelopment areas and tailoring federal grant schemes to local needs.

Considering different standards and construction years of buildings in stock, it is important to explore different scenarios, such as possible renovation versus demolition and new building or usage of standard versus sustainable material [3]. These calculations will take the embodied energy and possible reuse or recycling chances into account. In future scenarios, the renovation rate will be set higher, to create scenarios which meet the goal of a "nearly climate-neutral" building stock. It is currently projected that the retrofit goal will not be met with the current rate of renovation in existing buildings [16]. The example in Sect. 3 shows that in comparison to grid cell II, grid cell I both has high primary energy use and high embodied energy. Hence, renovation would help decrease the primary energy use, while preserving the embodied energy. From this small scale up to scales for cities or regions, evaluations can be done, which shows the spatial advantages of the tool (Fig. 7). In the next steps, the embodied energy of streets will be included in the calculation, to demonstrate dependencies between building arrangements and infrastructure construction.

It has to be kept in mind, though, that the tool is only an approximation. The census data is not precise and sometimes information is adjusted or omitted due to privacy protection. In addition, the customized data contains no information on dwelling sizes and vacancy rates, because each additional variable increases the likelihood of omission due to privacy protection measures. The classification is a simplification and there is no information on renovation rates and energy standards, which is why the explained assumptions have to be made (see Sect. 2.2). In order to improve the tool, Energy Performance Certificates (EnEV) should be made available, providing proper privacy protection. Regarding the LCA calculation, the results do not represent the exact amount of embodied energy, as, for example, the transportation of material was different in 1850 compared to present transportation impacts. Using the *Ökobaudat*

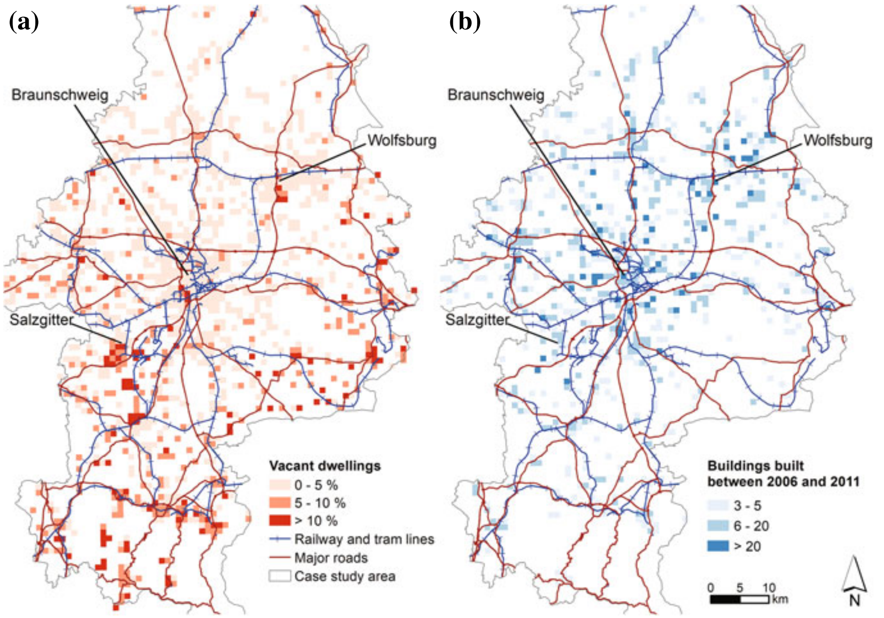

Fig. 7 Vacancy (**A**) and housing development (**B**) are both occurring in relative proximity in the eastern part of the case study area. This means that some embodied energy is not used, while additional energy flows into new building

2016 for the LCA assumes that the environmental impact of transportation is like it was 2016, not like 1850. Nevertheless, the overall impact and materials used are representative.

The full potential of the tool will become clear when combining it with other data, for example on mobility. In our example in Sect. 3, grid cell I is connected with a tram line that runs on a 10–15 min schedule during the day and connects it to the center of Braunschweig, while grid cell II only has bus service that runs less frequently and doesn't connect directly to the center of Braunschweig. Therefore, the need to use a car is probably much higher in grid cell II than in grid cell I. In fact, there are 1.3 cars per household in the municipality where grid cell II is located, but only 0.9 cars per household in Braunschweig, where grid cell I is located [9]. Unfortunately, there is no information on mobility on the grid scale, but estimations could be done through modeling.

The whole case study area has seen so-called expansive urbanization in the last decades, i.e. settlement and transportation areas have grown much stronger than the total population [8]. Some parts have even seen a declining population combined with an increase in settlement and transportation area. This has led to undesired outcomes, for example high rates of vacancy south and southeast of Braunschweig, in combination with low-density urbanization in the north and west (Fig. 7A, B).

Considering the embodied energy of buildings makes an even stronger case for the unsustainable nature of this diverging development.

In conclusion, the LCA tool is a promising instrument for regional sustainability assessments. Combined with other information, for example mobility analyses and environmental impacts of land consumption, it will allow exploring paths to greater sustainability of the built environment.

Acknowledgements This research was funded by the program "Science for Sustainable Development" of the Volkswagen Foundation and the Ministry for Science and Culture of Lower Saxony (metapolis, grant no. ZN3121). We would like to thank Clemens Schult from the Landesamt für Statistik Niedersachsen (LSN) for the customized census data and all our colleagues of the METAPOLIS team who contributed to our research.

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Part IV
Production and Logistics

Comparative Life Cycle Assessment Study on Cyanobacteria and Maize as Feedstock for Polylactic Acid

Maresa Bussa, Cordt Zollfrank and Hubert Röder

Abstract The move towards a bioeconomy requires to overcome the lack of biomass and to develop new processes for the production of chemicals and materials. Cyanobacteria can play a key role in the bioeconomy due to their fast growth and year-round production possibilities. In this study the life cycle assessment approach is applied in order to address three goals: (i) to evaluate the potential of cyanobacterial biomass as a replacement of maize as feedstock for polylactic acid; (ii) to identify the drivers of the environmental impacts; (iii) to assess three different improvement scenarios. Results show that cyanobacteria are currently not environmentally competitive with maize. The high electricity demand, the carbon dioxide requirements as well as urea are identified as crucial factor for the environmental impacts of cyanobacterial biomass. Replacing the electricity mix by wind power, reducing the carbon dioxide supply as well as upscaling of the lab-scale system reduces the environmental burden considerably. Further research is however necessary to optimize the production chain and to use biomass residues for valuable co-products.

Keywords Life cycle assessment · Cyanobacteria · Microalgae · PLA

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1 Introduction

In 2012, the European Commission announced the European bioeconomy strategy and action plan to promote the change towards a bio-based economy. The gap between the demand for and supply of biomass was identified as one of the main barriers to the economic transition [1]. Microalgae, including cyanobacteria, offer a potential for closing the gap between biomass demand and supply due to their exponential growth rates and the possibility of year-round production [2].

Cyanobacteria are recognized as a promising source for the high-value pharmaceutical market because of their primary and secondary metabolites [3]. Lipopeptides belong to the group of biosurfactants and are metabolites of special interest due to their antifungal and antibiotic activities. They could provide a solution for the increasing problem of multi-resistant bacteria as antibacterial agents based on lipopeptides have shown the potential to exhibit antimicrobial action against multi-resistant germs [4]. Furthermore, their antiadhesive properties enable lipopeptides for the application together with catheters and other medical insertional materials in order to slower the biofilm growth rate on the materials and thereby to reduce the number of hospital infections [5]. However, the share of lipopeptides in cyanobacterial biomass is low [3]. Hence, a large-scale production of lipopeptides would lead to high waste flows of the residual biomass. To foster the environmental and economic sustainability of lipopeptides from cyanobacteria, using the residue as feedstock for the production of a blend of polylactic acid (PLA) and cyanobacterial biomass is proposed.

PLA is expected to be the leading bio-based and biodegradable plastic. In 2014, the global bioplastics production capacity was 1.7 million tonnes, which is predicted to rise to 9.2 million tonnes by 2021 [6]. Thus, a sales market for a PLA-blend based on residual cyanobacteria biomass would be given. Presently, plants such as corn and potatoes serve as feedstock for the production of PLA. The properties of PLA are comparable to fossil-based polymers such as polypropylene (PP), poly-ethylene terephthalate (PET) as well as polystyrene (PS), resulting in a wide range of possible applications [7]. The main applications of PLA are packaging, textiles, consumer goods as well as agriculture and horticulture [6]. Moreover, its compatibility with the human body makes PLA an interesting material for medical applications [7].

The purpose of this paper is to investigate the potential reduction of environmental impacts of PLA by replacing a share of the PLA granulate based on terrestrial feedstock by cyanobacteria. Moreover, environmental hot spots of the algal biomass production are identified and improvement strategies analysed.

2 Methodology

2.1 System Boundaries and Functional Unit

Figure 1 shows on overview of the process chain, from the cultivation of the cyanobacteria to the production of the PLA-blend. This study focuses on the production of dry algal biomass, including cultivation, harvesting, cooling, disintegrating and drying due to present data availability. Cyanobacteria are cultivated in open raceway ponds and concentrated by centrifugation. The concentrated biomass is then cooled and disintegrated before the remaining water is removed in a spray dryer. Disintegration aims to fragment the cell structure of the biomass with the intention to increase the efficiency of downstream processes. Pumping between the different processes is included, whereas the required infrastructure for the system is excluded from the analysis due to lacking data. The location of the cultivation system is in the Czech Republic. The considered functional unit is 1 kg of dry biomass.

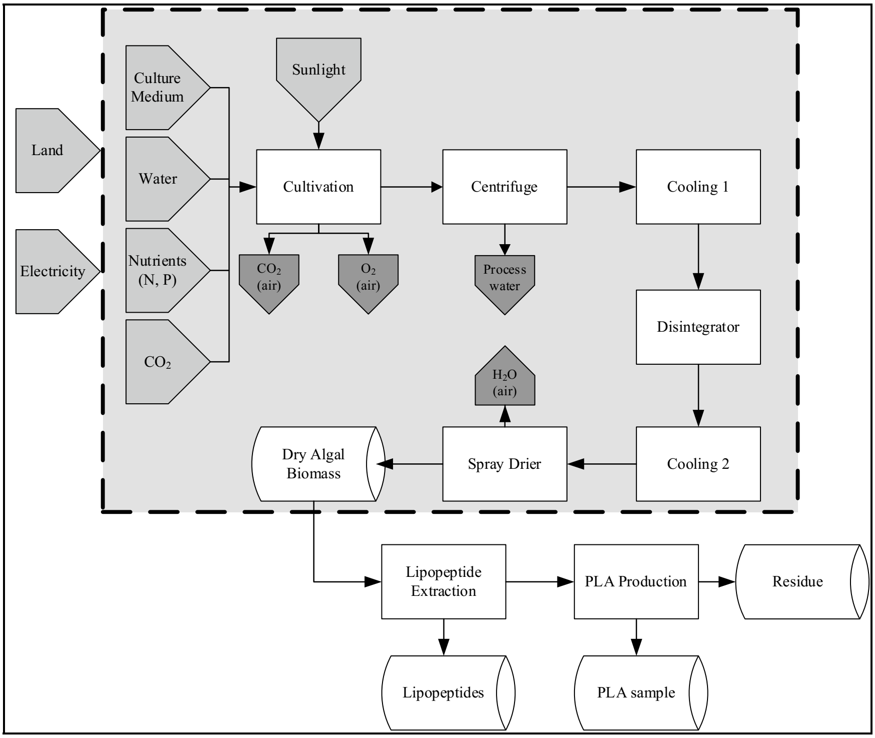

Fig. 1 System boundaries, analysed part highlighted in grey

2.2 Life Cycle Inventory

The data for the foreground system is based on calculations as well as measured lab-scale data from previous experiments with other microalgae strains. Ecoinvent v3.4 cut-off was used to model the background system. For water and electricity, country specific data for the Czech Republic was used, for wastewater treatment and CO_2 supply, average European data was used whereas global average data was used for the supply of nutrients.

The data set 'Maize grain {US}| production' of the Ecoinvent v3.4 database was selected as reference system since the main PLA producer is located in Nebraska and uses maize as feedstock. Thus, it is likely that cyanobacteria will replace part of the US maize as raw material for PLA. The data is based on literature data from 2006 and is extrapolated to 2017. The process system covers sowing, fertilisation, irrigation, application of pesticides, harvesting and transportation from the field to the farm as well as drying. Unlike as the cyanobacteria system required infrastructure such as machinery and sheds are included in the data set.

2.3 Life Cycle Impact Assessment

ReCiPe 2016 Midpoint v1.02 in the hierarchist version was used as life cycle impact assessment methods with SimaPro 8.5.2.0 since it is based on up-to-date modelling and covers a comprehensive set of impact categories. This study analyses all available impact categories in ReCiPe: global warming potential (GWP), ozone layer depletion (ODP), ionising radiation (IR), photochemical oxidant formation potential for human health (POFP-HH) and for ecosystems (POFP-ES), particulate matter formation (PMF), terrestrial acidification (TAP), freshwater eutrophication (FEP), marine eutrophication (MEP), human carcinogenic toxicity (HTP-c), human non-carcinogenic toxicity (HTP-nc), land use (LU), mineral resource depletion (MRD), fossil resource depletion (FRD) and water consumption (WC).

2.4 Improvement Scenarios

A contribution analysis for the baseline scenario described before was conducted to identify the main processes responsible for the environmental burden of the product system, which are equal to the most promising levers for improving the environmental performance of the system. Based on the results of the contribution analysis three different improvement scenarios were proposed:

- Scenario 1: wind energy
- Scenario 2: scenario 1 plus lower CO_2 supply
- Scenario 3: scenario 2 plus lower electricity demand.

For scenario 1, the Czech electricity mix is replaced by wind energy generated by small-scale (<1 MW) onshore wind power plants. Therefore, the electricity mix dataset in Ecoinvent was adjusted, whereby transmission and transformation impacts are assumed equal.

For scenario 2, consulted experts proposed a reduced CO_2 supply by 60%. The reduced supply also leads to less excess CO_2 emitted into air.

For scenario 3, the most electricity demanding processes were identified based on the inventory data. As pumping the biomass through the open raceway ponds and spray drying are together causing more than 90% of the electricity demand, their electricity consumption was adjusted based on literature values for large-scale facilities. According to Chisti [8] the usual energy demand of large-scale raceway ponds is in the range of 0.5–1.5 W/m^3. The upper value was assumed for the daytime and the lower for night-time. The electricity demand of the spray drier was recalculated based on the specific energy consumption of 3 GJ/t water evaporated [9].

3 Results

3.1 Baseline Scenario

Figure 2 shows the comparison between cyanobacteria and maize. Cyanobacteria are clearly outperformed in all impact categories. On average, the impacts of maize cultivation are 94% lower.

The heavy electricity demand of the cyanobacteria process was identified as main driving force for the environmental impacts. It contributes with 88–97% to all impact categories, except for mineral depletion and terrestrial ecotoxicity, where the contribution is 67 and 54% respectively. For both impact categories, the carbon dioxide

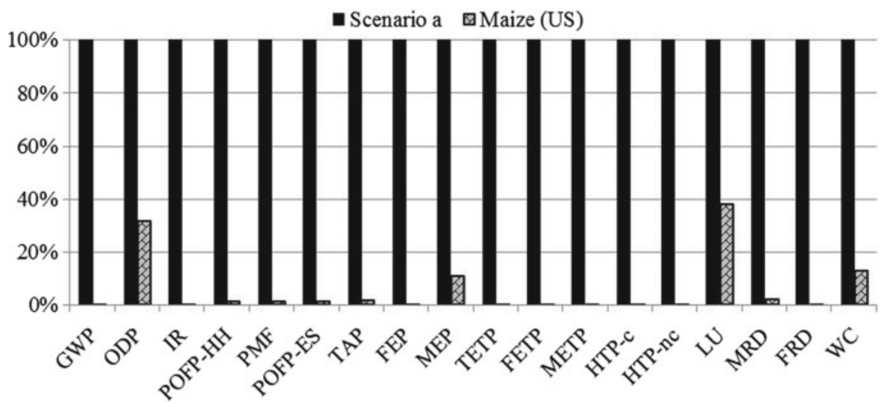

Fig. 2 Comparison of cyanobacteria and maize (baseline scenario)

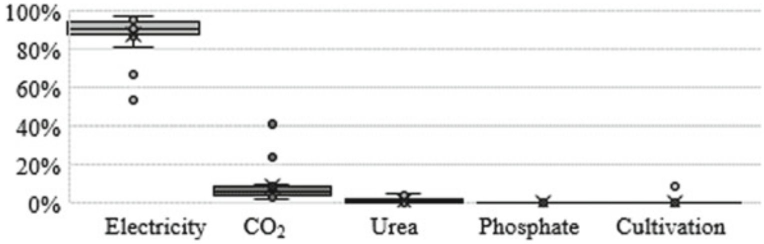

Fig. 3 Contribution analysis for the baseline scenario

supply is important as well, whereby it plays a tangential role for the remaining impact categories. The impacts of urea and phosphate are negligible compared to the influence of electricity and carbon dioxide demand. The cultivation stage contributes with 8% to the global warming potential due to the release of the excess carbon dioxide into the air (see Fig. 3).

3.2 Scenario 1

Replacing the Czech electricity mix by wind energy reduced the impact by 72% on average in all impact categories except for mineral resource depletion, where the impact increased by 37%. For terrestrial ecotoxicity, the reduction of 0.5% is marginal. Cyanobacteria cultivated with wind energy has lower land use impacts than maize and the marine eutrophication potentials as well as the water consumption are in the same order of magnitude (see Fig. 4).

As can be seen in Fig. 5 the electricity demand is still the most important driver for the environmental performance of cyanobacterial biomass. However, its contribution decreased considerable for most impact categories except for terrestrial, freshwater

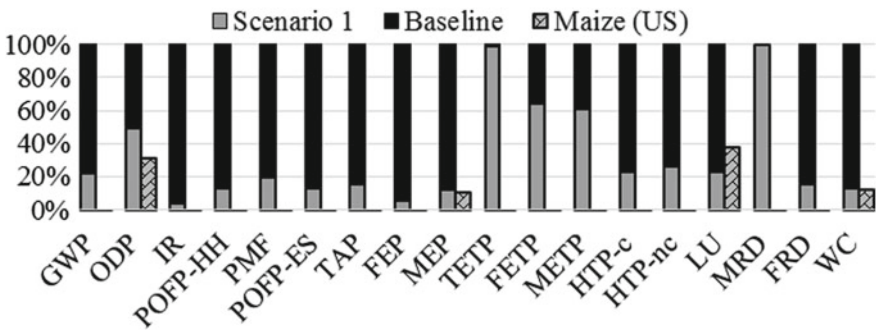

Fig. 4 Comparison of cyanobacteria and maize with wind energy (scenario 1)

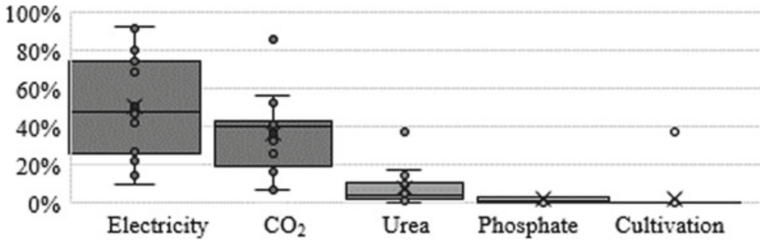

Fig. 5 Contribution analysis for scenario 1

and marine ecotoxicity, where the contribution remains almost equal, and mineral depletion, where the contribution increased by nine percentage points. These results are due to the production chain for wind power plants: Copper production is causing 40% of the terrestrial ecotoxicity potential, whereas the treatment of scrap copper is responsible for around three-quarter of freshwater (77%) and marine ecotoxicity (74%). Furthermore, sulfidic tailing is causing human non-carcinogenic toxicity (60%) and freshwater eutrophication impacts (39%). The carbon dioxide supply is increasing in importance and responsible for 20–40% of the environmental damage in most impact categories. Especially ionising radiation is mainly driven by the carbon dioxide demand (87%). Urea has a median of 4% and is with 37% the main water consuming substance. The use of phosphate fertilizer plays only a minor role causing a maximum of 4% of the impacts.

3.3 Scenario 2

Reducing the supply of carbon dioxide leads to an additional improvement of 6% on average in all impact categories. Especially for terrestrial ecotoxicity, global warming and mineral resource depletion considerable reductions are achieved. With wind power and reduced carbon dioxide supply cyanobacteria outperform maize in land use, water consumption and marine eutrophication as shown in Fig. 6.

Reducing the carbon dioxide supply by 60% almost halve the contribution of the cultivation stage to the global warming potential (from 38 to 20%). It reduces the contribution of the carbon dioxide production chain by 4–22 percentage points. This leads to an increasing important of the electricity demand for the environmental performance of the system. Urea is with a contribution of 46% the main driver for water consumption and a non-negligible factor for fossil resource depletion (27%), terrestrial acidification (19%), particulate matter formation (16%) as well as photochemical ozone formation (14%) (see Fig. 7).

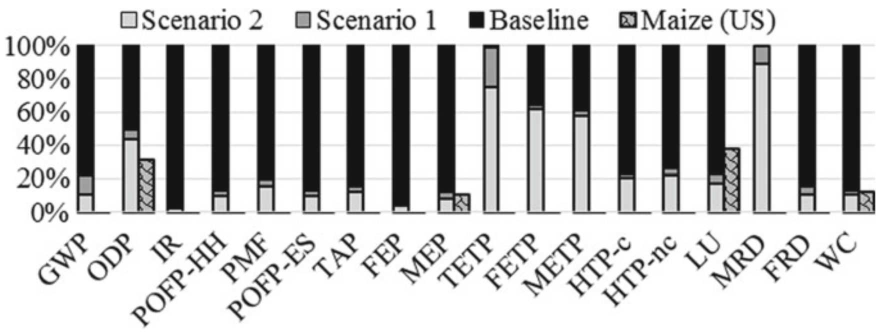

Fig. 6 Comparison of cyanobacteria and maize with wind energy and lower CO_2 supply (scenario 2)

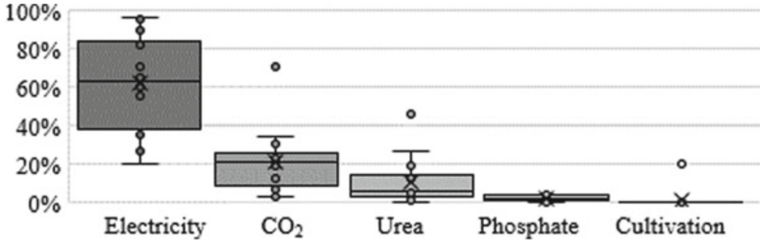

Fig. 7 Contribution analysis for scenario 2

3.4 Scenario 3

Upscaling of the two main energy-consuming processes reduced the environmental impacts on average by 18%. Especially for mineral resource depletion (67%), freshwater ecotoxicity (53%), marine ecotoxicity (49%), terrestrial ecotoxicity (48%) and ozone depletion potential (33%) significant environmental gains were achieved. Compared to maize, cyanobacteria are advantageous in terms of land use, marine eutrophication, water consumption and ozone layer depletion. For freshwater eutrophication and ionising radiation the results reach the same order of magnitude (see Fig. 8).

As can be seen in Fig. 9, carbon dioxide is the main driver for most impact categories in scenario 3. The release of the excess CO_2 in the cultivation stage is causing 28% of the global warming potential. The impacts of the energy reduces considerably, however, it remains the main factor for freshwater and marine ecotoxicity as well as for human carcinogenic toxicity potential. These impacts are mainly caused by the treatment of scrap copper and slag from electric arc furnace steel production taking place in the wind farm supply chain.

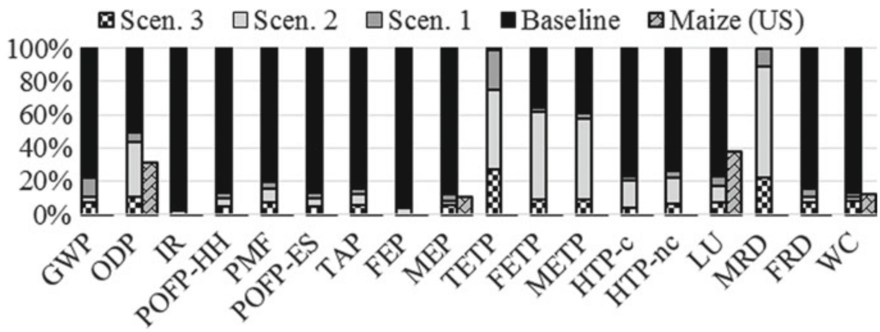

Fig. 8 Comparison of cyanobacteria and maize with wind energy, lower CO_2 supply and up-scaled (scenario 3)

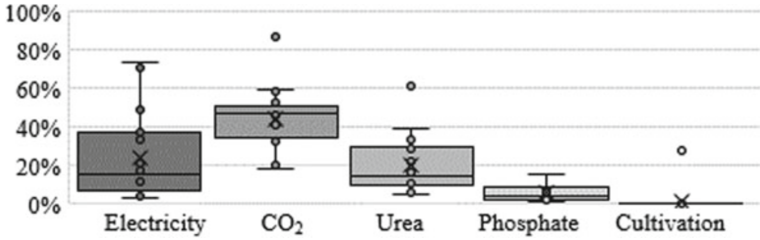

Fig. 9 Contribution analysis for scenario 3

4 Discussion

The study is facing few limitations that needs to be taken into account before drawing conclusions. Firstly, as described in Sect. 2.2 the system boundaries of both compared systems differ due to data availability. No infrastructure is included in the cyanobacteria system, leading to advantages in the comparison with maize. Secondly, both compared systems differ in scale and maturity. The cyanobacteria process does not exist at this stage at industrial scale and many technological problems are still unsolved. Whereas the maize dataset is based on a mature technology and large-scale cultivation. As a higher maturity and scale of a system leads to more efficient use of resources, maize has an advantage over cyanobacteria. Thirdly, the functional unit of 1 kg of biomass does not consider which share of the biomass is eventually used in the PLA-blend. The correct functional unit for comparing feedstocks for PLA-blends would be one 1 kg of PLA-blend with equal properties. However, as downstream processes from the dried cyanobacterial biomass to the PLA sample are currently under investigation and data is lacking, 1 kg dry biomass is the best applicable functional unit at this stage. This study must therefore be interpreted as a LCA driven study identifying environmental hotspots of the cyanobacteria process. The main objectives of the study are to identify the parameters, which have the most impact on the environmental performance of the system, and to analyse the environmental effects of improvement options based on currently available technology.

The contribution analysis showed that the energy demand, carbon dioxide requirements and urea are the main driving forces of the environmental performance. Especially the energy demand and the energy mix were identified as crucial factors for the environmental sustainability of the system. The analysed scenarios suggest that promising improvement options based on current technology are available and that the environmental performance of cyanobacterial biomass can be improved significantly in the mid-term future. However, further research is necessary to be environmentally competitive with maize. Especially the human non-cariogenic toxicity (64 times higher than for maize), marine ecotoxicity (44 times higher), terrestrial ecotoxicity (42 times higher) and freshwater ecotoxicity potential (33 times higher) need to be reduced. As these impact categories are mainly driven by the production chain of wind power plants, further studies on the most suitable and realistic energy mix for cyanobacteria cultivation facilities are required as well as further effort to reduce the energy demand of the system. Furthermore, other possible improvement scenarios were not considered in this study: closing the loop of the process water or using waste or flue gas streams for the provision of nutrients and carbon dioxide could significantly reduce the environmental impacts.

5 Conclusion

Despite their potential cyanobacteria are not yet environmentally competitive with maize. Energy requirements, carbon dioxide demand and, to some extent, urea have been identified as main drivers of the environmental impacts. With wind power, reduced carbon dioxide supply and lower electricity demand the impacts of cyanobacterial biomass could be reduced by 90% on average in all impact categories. In that case, cyanobacteria outperformed maize in land use, marine eutrophication, water consumption and ozone layer depletion. However, in other categories like human non-carcinogenic toxicity as well as marine, terrestrial and freshwater ecotoxicity the values for cyanobacteria compared to maize are many times higher. These impact categories are mainly driven by the production chain of wind power plants. Hence, for an environmental sustainable cultivation of cyanobacteria further research on an optimum and realistic energy mix is needed to avoid burden shifting from one category to another. Furthermore, it is recommended to investigate strategies for reducing the electricity demand and to assess alternative sources for carbon dioxide and nutrients. The development of a sound zero-waste biorefinery concept for cyanobacteria biomass is a critical factor to reduce the environmental burden allocated to individual outputs such as the PLA-blend.

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Assessment of Sustainability on Dairy Farms in Central Germany Based on Energy and Nutrient Balances

Clara Heider-van-Diepen

Abstract An ecological evaluation of farms is shown using "REPRO", a software, which makes it possible to represent agricultural actions including consequences. The system approach reduces the complexity of nature to a cycle where agricultural activity has an influence on. Thus, all decisions play a significant role. "REPRO" enables e.g. a condition analysis. The program consists of a crop cultivation and an animal area. For the overall farm analyze, the crop production results are transferred to the animal area. The climate impact is illustrated by a greenhouse gas balance, the ratio of nutrient emissions and energy intensity in their carbon dioxide equivalents to the product. The results show to what extend the examined dairy farms differ and where they resemble each other. It is shown that keeping cattle with an increased amount of straw and solid manure disposal leads to an increase in nitrous oxide emissions compared to low straw stabling and liquid manure removal. But in slurry storage this leads to a higher methane emission. Oversizing of agricultural machinery in animal feed production could also be identified. The ratios to the product show a high degree of similarity in two farms, although one farm has significantly lower emissions.

Keywords Sustainability · Dairy production · REPRO · Energy and nutrient balance · Carbon footprint

1 Motivation

"From its beginnings in economics and ecological thinking, sustainability has become a planning concept and has been widely applied in rural development" [1].

One of the greatest challenges of our time is to ensure that society's needs are met without burdening future generations with environmental interventions in nature

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[2]. Agriculture is facing this challenge to a particularly large extent, as the steadily increasing population and the constantly decreasing fossil raw material reserves require a rethink [3].

Sustainable thinking was initiated at the United Nations Conference on Environment and Development Policy in Rio de Janeiro in June 1992. A new mission statement, the “Agenda 21” action plan, was drawn up for international environment and development policy. It calls for the development and application of indicators for the evaluation and analysis of sustainable management [4].

Since no uniform definition of sustainable management is known to date, various projects and studies are attempting to develop initial approaches for a sustainability assessment. The “REPRO” model provides an evaluation approach for agricultural action. The software “REPRO” is a tool for environmental management in which the management of agricultural land can be presented on the basis of nutrient and energy balances.

2 Systems Approach

Many species, including those hardly related to each other, form life communities (biocenoses) in various forms. These communities are dependent on inanimate environmental influences (biotopes), i.e. air, water, light, temperature and nutrient content, and interact to form an ecosystem. This ecosystem forms a continuous cycle whose material movements are the basic idea for the system approach.

The system approach used for the analysis of agricultural holdings reduces the complexity of nature to a (material and energetic) cycle with the compartments “soil”, “plant” and “animal”. An agricultural land use creates a direct relationship between all of them, which can be influenced by action. Figure 1 shows this cycle schematically.

The three compartments and various (exemplary) influencing factors can be seen. The characteristics of the seed and the crop rotation, the choice of nitrogen fixers and the cultivation of humus multipliers or humus-suppressing plants, for example, have a great influence on the plant and its growing process. In addition, the choice of machinery has a contribution to the energy balance because weight, working width and power influence the energy consumption per hectare and cause more (or less) pressure on the soil structure. The harvest products are carriers of the nutrients, which leave the cycle as output quantities or serve for the following compartment. The animal area is clearly defined on the one hand by the choice of the farm animal species and on the other hand by the choice of the type of husbandry. Many of the input variables in this area are already defined by the actions and decisions in crop production. In the animal production sector, influence can be exerted in the composition of the feed rations and the form of husbandry as well as the storage of manure. At this point, nutrients leave the cycle in the form of production goods (milk, meat) or emissions [methane (CH_4), nitrous oxide (N_2O) or ammonia (NH_3)].

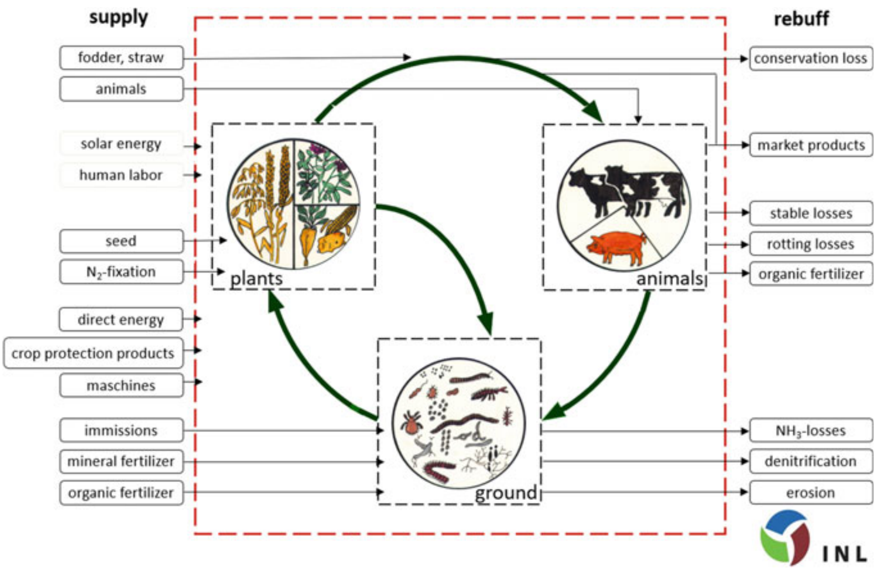

Fig. 1 Mapping of the farm as a networked system

The remaining nutrients are passed on to the soil via the excrements and thus to the closed-circuit compartment.

Although the soil is clearly defined by its structure, the various options for action can have numerous effects. The choice of fertilizers (mineral/organic), pesticides (intensity and product) or the type of stabling used (slatted floor, bedding) that cause emissions, nitrogen losses and other external impacts.

Through “REPRO”, there is now a possibility to present this material and energy cycle and thus to estimate the consequences of agricultural activity.

3 Software

“REPRO” (Reproduction of Soil Fertility) is a computer-aided balancing software developed since 1996 in an interdisciplinary manner in research projects of the Martin-Luther-University Halle-Wittenberg. It implements system- and process-oriented analysis and evaluation approaches on the basis of indicators. The indicators are related to each other and are not only mapped isolated. Material and energy flows map the interactions between the subsystems (crop production, animal husbandry, arable land, grassland) within the system boundary of a farm. Thus, the complexity of agricultural operating systems is taken into account. The model approach makes it possible to carry out scenario calculations with possible target states in addition to actual situations.

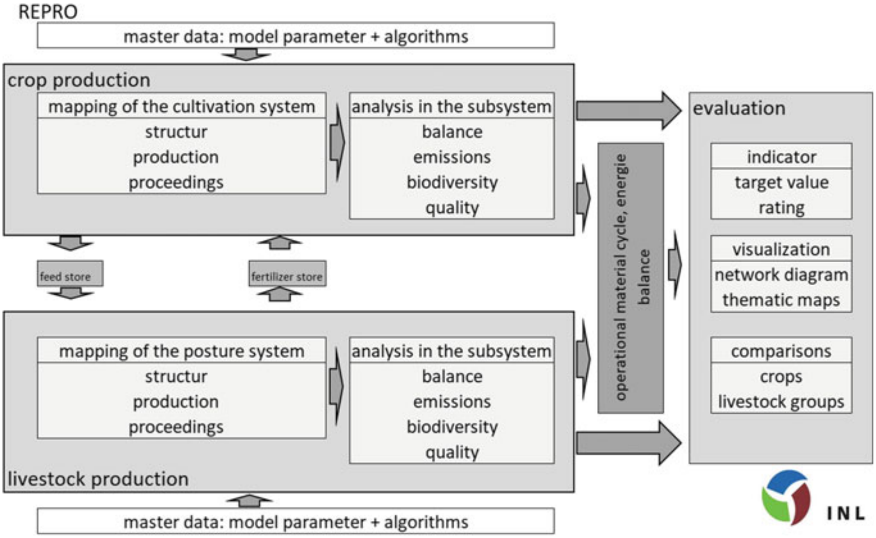

Fig. 2 Schematic structure of a complete farm within "REPRO" incl. networking of animal husbandry and farm cultivation

Farms are mapped as an overall system by defining individual subareas of the farm (location, crop production, livestock production) as subsystems, adaptable via modules, and linked to each other. Figure 2 shows the schematic representation of a networked farm.

The reality is shown quite exactly here. In the data acquisition essential information about the management system is recorded and applied yearly on stable area/animal group level or in plant production on (partial) field level. In crop cultivation, in addition to all fixed parameters such as soil type, number of fields, field sizes and soil examinations, variable action parameters are also recorded. This includes all measures of a business year from soil cultivation, sowing, fertilization and plant protection to harvesting including the machines and means used (type and quantities). By means of these individual farm input data and with the help of model parameters and algorithms, the emission-relevant output variables are calculated for the procedures and allocated to the individual sublots and harvests. The animal sector is a separate module complex containing the mapping and calculation of the specific indicators. The husbandry method is recorded in detail by specifying individual farm housing systems, stock and performance parameters as well as feeding methods.

Material and energy balances are drawn up from the calculation of input (feed, inputs, animals) and output (products, manure) quantities and the efficiency and consumption of resources are presented. Farm networks and interactions are taken into account through consistent, overlapping data collection. The feed store and the manure serve as an interface between farm cultivation and animal husbandry, making it possible to map internal material cycles.

4 Methodology

Sustainability is assessed on the base of indicators. Indicators provide quantitative and qualitative information on the condition and development of complex systems. Environmental indicators of agricultural landscapes allow statements to be made about the sustainability of land use systems, their influence on contiguous ecosystems and changes in their temporal and spatial dimensions. In this way it is possible to simplify complex conditions and to estimate their consequences in order to present information for decision support.

4.1 Crop Production

Various indicators can be identified in crop production. These include, among others, humus and nutrient balances, biodiversity and active substance intensities as well as energy and greenhouse gas balances.

The exact calculation methodologies will not be discussed in detail here, but can be found in various sources like Hülsbergen 2002 or Küstermann 2008 [5, 6]. At this point, the relationship between the indicators humus balance and greenhouse gases will be shortly explained as a representative of the crop cultivation indicators and the cycle-related interdependencies of the indicators on each other.

Figure 3 shows that the carbon dioxide (CO_2) potential is significantly lower with a positive humus balance than with humus-rich cultivation systems. In order to calculate the CO_2 quantities for the greenhouse gas balance in the crop production, it is necessary to analyze the relevant nitrogen (N), carbon (C) and energy fluxes as a function of location and management conditions. The C storage or release in humus is determined using the humus balance (dynamic approach). A positive humus balance binds carbon in the soil. This reduces the carbon pool available for CO_2 formation in the atmosphere. Depending on the extent of the humus balance, a positive or negative balance takes place in the calculation of the greenhouse gas balance. It is therefore a mathematical credit that reduces the greenhouse gas balance if the humus balance is positive.

As already described, the calculated indicator values are automatically passed on to the animal area. The results of the plant production are not further described at this point but can be seen in Heider-van Diepen 2018 [7].

4.2 Animal Area

The focus of this study is on the animal sector. For this reason, the indicators used will be outlined here and the summary indicator of the greenhouse gas balance will be explained in detail.

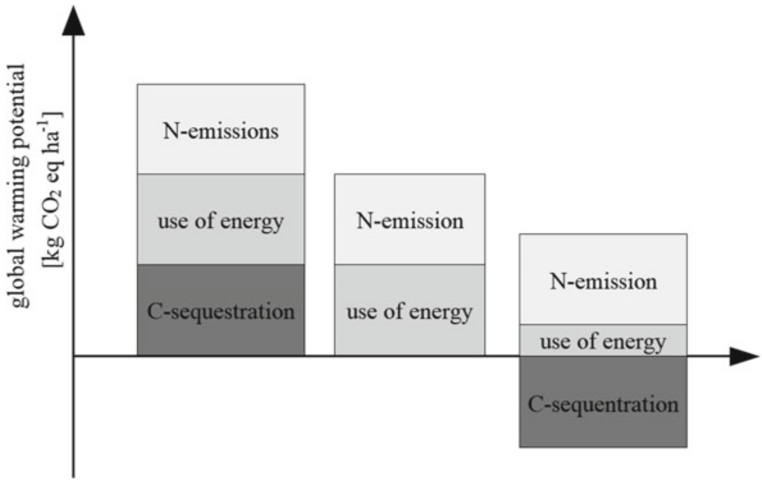

Fig. 3 Greenhouse gas balance—methodology

The animal sector was assessed in terms of nutrient efficiency [nitrogen and phosphorus (P)], nutrient emissions (methane, ammonia and nitrous oxide) as well as energy and greenhouse gas balance.

Nutrient efficiency consists of nitrogen and phosphorus efficiency. The nitrogen/phosphorus efficiency represents the ratio of the N/P input via the feed to the product produced, based on the nitrogen/phosphorus cycle of the farm. Nitrogen/phosphorus compounds, which are absorbed via the feed, are partly used by the animal for the synthesis of body tissue or milk proteins. The lower this ratio, the better the nutrient efficiency. This ratio is compared with a performance-related, farm-specific target value, which is calculated on the basis of "good professional practice" values. Neither an underrun nor an overrun would speak in favor of a sustainable management of the farm in terms of nitrogen and phosphorus use.

Nutrient emissions are calculated in real quantities in the form of methane, ammonia and nitrous oxide. The methane emission indicator reflects methane emissions from fermentation in the rumen of animals (enteric) and during the storage of manure. Only an approximate estimate can be made, as the formation of methane in the rumen is determined by various influencing factors. Various regression equations exist for this purpose. Decisive factors in the calculation include gross energy consumption, crude fiber content, crude protein content or crude fat content in animal feed [8, 9]. In the farmyard manure stores, in addition to the structural facilities, the dry matter uptake and the excreted organic matter, the residence times in stables and pastures with the respective type of manure are also taken into account [10].

For the ammonia emission the composition and storage of the animal excrements is of special importance. These are the causes for the extent of the chemical transformation processes and thus decisive for the level of emissions. In particular, the concentrations of inorganic carbon, ammoniacal nitrogen, organic acids and organic components determine the reactivity on the surface of the stable floor and during storage [11, 12].

The nitrous oxide emissions also come not only directly from animal production, but also from the conversion processes of manure during storage and after application to the field. These microbial processes are largely dependent on the external conditions on site and the bacterial population in the mass. The environments with a low oxygen content in which both aerobic and anaerobic conditions occur are favorable. For quantification, all organically bound N-sources as well as Total Amminical Nitrogen and storage-specific emission factors are used [10].

The energy intensity reflects the general efficiency with energy expenditures of any kind. In this indicator, the product-related energy input is compared with the generated product. Energy input is examined in connection with the farm structure (direction of production, animal stocking), the intensity level (use of direct energy), the material balance (feed use), the process design (buildings and structural facilities) and the performance level. It is necessary to record the direct use of energy in the form of fuel and power as well as electricity within the system boundary. Indirect energy use is ensured by internalization of the energy expenditure for production-relevant factors required for a farm. The basis for the calculation is provided by the process analysis, in which the energy expenditure of the individual process steps (feed production, manure and feed storage, husbandry system, offspring breeding, milk production, pasture, machinery and technical equipment) is allocated to direct and indirect energy flows. Energy costs for direct feed supply, transport to the farm or onward transport from the farm, solar energy and human labor are not included. The energy expenditure for the provision of animal feed is precisely determined by coupling it with the "REPRO" model via the mapping of crop production impact.

The greenhouse gas indicator relates the climate-relevant emissions of a company to the product. In addition to the indicators mentioned for milk-producing cattle, the overall farm result for livestock farming also includes the emissions caused by offspring. In the animal sector, the produced edible protein (eP) is selected as the reference value. It is a cumulative indicator whose representation takes into account the process-related CO_2 emissions from energy input, the direct and indirect (from NH_3) N_2O and CH_4 emissions from cattle farming. The respective sub-elements are converted with their specific carbon dioxide equivalents (CO_2 eq) and summed up. The carbon dioxide equivalents reflect the respective global warming potential. They are assumed after IPCC 2006 [13] as follows:

- methane 25 kg CO_2 eq
- nitrous oxide 298 kg CO_2 eq
- ammonia 3.86 kg CO_2 eq
- energy 0.152 kg CO_2 eq.

Finally, the calculated CO_2 emissions are set in relation to the edible protein produced. The edible protein is a cross-product reference level that refers to the protein contained in the product. This is used to ensure comparisons between different production directions. For this purpose, the values from the results of Flachowsky 2011 [14] are standardized. Accordingly, 95% of the protein in the milk is counted as edible and a usable proportion of 50% is assumed for the quantity of meat.

5 Results

The results reflect the overall results of the research farms. These objects of investigation represent three dairy farms in Central Germany. In order to differentiate between the farms, they were designated as farm A, farm B and farm C.

5.1 Result Chart

The carbon dioxide equivalents of the individual balance members of the greenhouse gas balance are compared. Figure 4 shows the results of the greenhouse gas balance from the various farms. The bars are divided into the balance members of the offspring, ammonia emission, nitrous oxide emission, methane emission (enteric, manure stock) and energy intensity. The overall farm efficiency can be seen in the form of the cross.

The overall operational results indicate that the output quantities of farm B and farm C are at a similar level while those of farm A are rated a little underneath. The

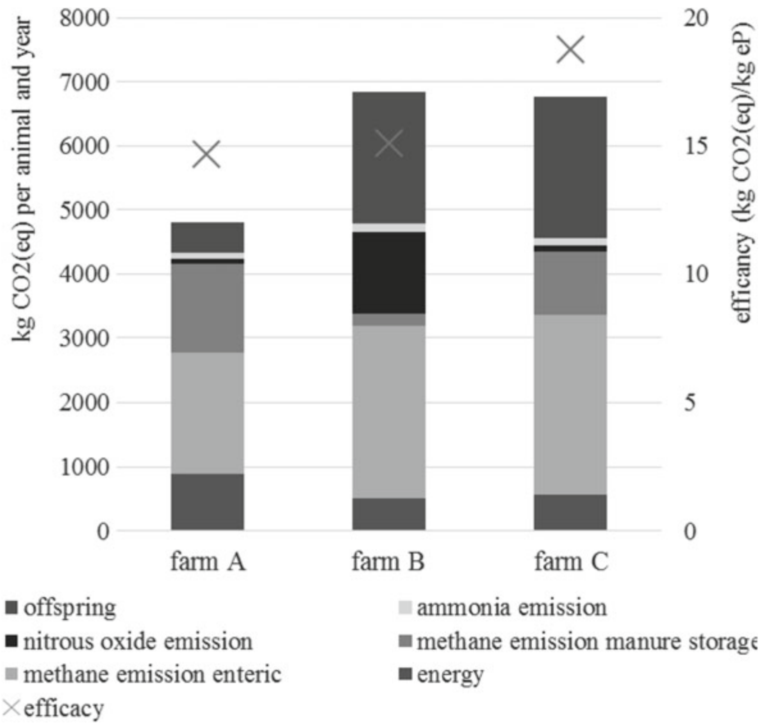

Fig. 4 Evaluation of the indicator greenhouse gases in the research farms

same ratio can be seen for the output quantities in the form of enteric methane. In the case of emissions in the form of methane coming from the storage, the quantities produced in the farm manures show large differences, with the value for farm B far below that of farm A and C. The CO_2 eq output for farm A is more than 1 ton above that of farm B. Concerning the evaluation of the energy expenditures in kg CO_2 eq, the result of farm A is significantly higher than that of farm B and C. A striking feature of nitrogen losses in the form of nitrous oxide is the similar level of farm A and farm C (0.14 and 0.2 kg $cattle^{-1}a^{-1}$). At 2.7 kg per dairy cattle per year, on the other hand, farm B emits significantly larger quantities of laughing gas per year. Ammonia emissions differ only by a maximum of 88 kg CO_2 eq per year in all farms.

In the overall view of the emission created by the companies, the maximum difference is 2.5 t CO_2 eq. This difference is particularly marked by the emissions generated by the offspring. These emissions are significantly lower in farm A than in farms B and C. Without offspring, all farms would be at a similar level (maximum difference 400 kg). With the given results, farm A shows the lowest emissions per dairy cattle, the other farms are on almost the same level, but significantly higher.

If one considers the ratio to the kilogram edible protein (kg eP) produced (farm A: 289,82 kg eP, farm B: 315,62 kg eP, farm C: 252,76 kg eP), the efficiency is highest in farm A and lowest in farm C. The efficiency of farm B is at the same level as that of farm A despite of much higher emission output due to the higher output level.

5.2 Interpretation of the Results

The results of the individual companies can be explained on the basis of the farm structures. The emissions caused by the offspring are so low in holding A, for example, because the holding sells calves at a young age and buys back occupied heifers (an improvement in the standard program values is aimed at here). The differences in nitrous oxide emissions are due to the type of farming. In farm B, the playpen is generously bedded with a lot of straw and most of the manure is stored as solid manure. In this form, there is a higher nitrous oxide emission potential than with liquid manure storage as can be found in farm A or C. The nitrous oxide emission potential is higher in this form than with liquid manure storage as can be found in farm A or C. The nitrous oxide emission potential is higher in this form than with liquid manure storage as can be found in farm A or C. On the other hand, methane emissions from farm manure in farm B are much lower than in the other farms. The amount of methane from the storage facilities is additionally depending on the digestibility of the feed rations of the animals. The less the animals are able to digest the feed, the more organic matter remains in the excrements and reactions to be emitted. The methane emission directly from the animals is lower the more ruminant the feeding is designed. In the balance element of the energy evaluation, a more detailed analysis of farm A revealed a significant overcalling of the machine use in feed production.

Finally, it should be pointed out that the higher production output (edible protein) of farm B allows the same product-related efficiency to be achieved as in farm A despite slightly higher quantities of emissions. This shows the comparability through the reference value, the production output, which would not be possible without it.

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Carbon Footprint Accounting for General Goods—A Comparison

Daniel Hülemeyer and Dustin Schoeder

Abstract Carbon emissions are an actual topic in the European public discussion due to the climate objectives of the European Union and the countries in specific. Especially the traffic and the transport sector have been evaluated as polluters so that logistics service providers and forwarders are getting into the focus of their customers to provide a systemic approach in calculating the carbon footprint of their logistics services. In addition customers struggle to compare carbon footprints of several logistics service providers and forwarders, due to the lack of a common standard for calculating of carbon emissions. There are two existing approaches, which are getting in focus for the logistics companies if they have not outsourced the calculation or are using fee-based calculation software. On the one hand, there is the European standard EN 16258, which has been established in Europe to create more transparency and on the other hand there is the GLEC framework as a global approach which aims to create a global standardized procedure. Both standards allow different approaches for emission calculation which lead to totally different results of carbon emission. Due to that a comparable value for customers and stakeholder is not given.

Keywords Carbon footprint · Logistics · EN 16258 · GLEC framework

1 Introduction

Carbon emissions are an actual topic in the European public discussion due to the climate objectives of the European Union and the countries in specific. Especially the traffic and the transport sector have been evaluated as polluters so that logistics service providers and forwarders are getting into the focus of their customers to provide a systemic approach in calculating the carbon footprint of their logistics services.

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In addition customers struggle to compare carbon footprints of several logistics service providers and forwarders, due to the lack of a common standard for calculating of carbon emissions.

Therefore a valid database and common approach is needed, so that logistics service providers and forwarders are able to calculate transport related emissions on a common basis and provide comparable data in their sustainability reports and towards the customer. Especially the calculation of carbon footprints in transportation of general goods needs a lot of data about the different transport chains and participants in it.

There are two existing approaches, which are getting in focus for the logistics companies if they haven't outsourced the calculation or are using fee-based calculation software. On the one hand, there is the European standard EN 16258, which has been established in Europe to create more transparency and on the other hand there is the GLEC framework as a global approach which aims to create a global standardized procedure. Both standards shall be compared in the following paper.

2 Footprint Accounting Practices

In recent years different regulations and guidelines have been set up in order to support companies in calculating transport emissions. With the EN 16258 a European standard for emission calculation has been published in May 2013. With this standard the formal requirements of the Greenhouse Gas Protocol have been transferred on transport services and have become a framework for logistic providers.

The toolset of the "DSLV Guideline" is a recommendation for the use of the standard, furthermore it is more precise than the EN 16258.

With the GLEC Framework a supplementary guidance for emission reporting according to the Greenhouse Gas protocol has been created for the logistics sector to encourage voluntary emission reporting [1].

2.1 EN 16258

The objective of the "Methodology for calculation and declaration of energy consumption and GHG emissions of transport services" (EN 16258) is the creation of a framework as a general approach for the determination of carbon emission in transport service [2].

In Scope of this standard are fuel and energy consumption with their specific emissions of greenhouse gases as well as the upstream emission of degradation, refining, transport and distribution ("Well to Wheel"). Emissions out of spills or by short-time support (e.g. tugboat) are out of scope. External handling as well as facility related emissions and energy consumption of IT stay disregarded, too [2].

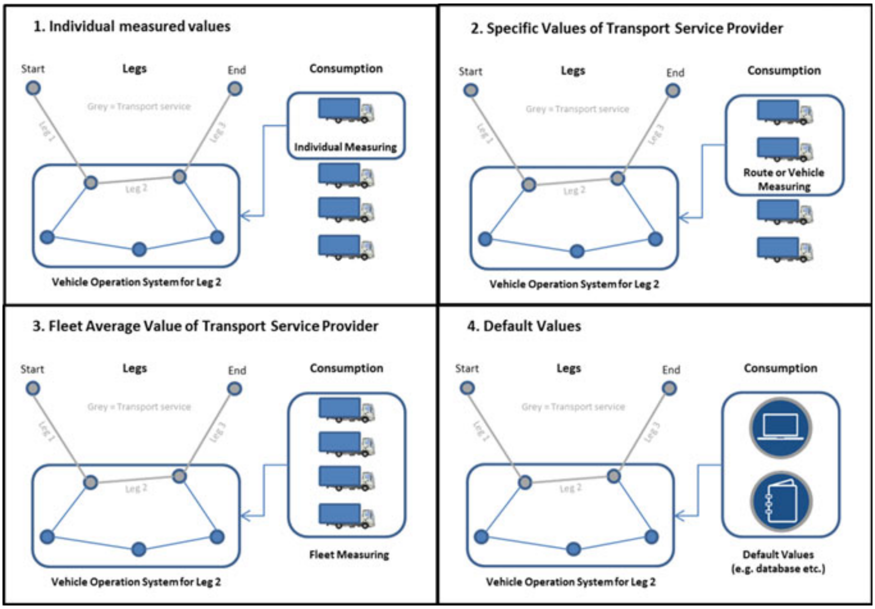

Fig. 1 Different varieties of emission calculation of EN 16258. *Source* Own representation based on [3]

The EN 16258 defines four different ways for the calculation of transport emissions. Preferred source should be—following option one—the measuring of carbon footprint emissions of every single transport. Subsequently this value is spread on all transported shipments according to their weight and their distance by using the unit ton-kilometers (t km). In case of using subcontractors this method becomes a huge challenge for the logistics providers as data are often not available or data quality is questionable. Moreover, an evaluation following this method requires quite a lot of personal and financial resources as well as data interfaces between the vehicles and the transport management system (TMS), which have to be implemented.

Second option is to evaluate the average fuel consumption for specific routes or vehicles. Due to the above mentioned problem of data quality in subcontractor operated business, the second option is also difficult to implement.

As third option, if the first and second are impossible to conduct, the standard recommends to utilize an average value of the fleet. In transportation of general goods the own fleet, the subcontractor's fleet and the partner's fleet are heterogeneous and an exchange of fuel consumption data is uncommon. In combination with the already mentioned data quality problems a fleet average value can be inconsistent (Fig. 1).

The weakest method of the standard, but the most common alternative, is the application of default values based on the unit ton kilometers, which are provided by different guidelines and reporting tools.

$$Ton\,kilometers = Weight\,in\,t * Distance\,in\,km$$

Even if this is a practical approach, it is most inaccurate, because investments in a modern motor pool or driver efficiency trainings will not affect the calculated result. For the customer a comparison of two transport providers covering the same route with the same type of vehicle will give no sufficient result as the result of carbon footprint evaluation—following option four—will be identical, if they use the same input factors.

2.2 DSLV Guideline

The guideline of the German Association for transportation and logistics (DSLV) describes the practical use of the EN 16258. It helps to identify the requirements of the standard [4].

With a variety of standard emission factors for different modes of transport and transport vehicles the guideline offers a lot of additional benefit regarding the evaluation of carbon footprint emissions. Especially the possibilities to analyze general goods transport and allocate the emissions to the single shipments are clarified. Furthermore, this guideline offers a possibility to include emissions of cargo handling and emissions caused by buildings, IT or warehouse forklifts. However, an allocation of these indirect emissions on shipment level linked to the weight of shipments or the number of handled packages is rudimentary explained and a single example is given. Nevertheless the main focus of the guideline is the transportation of shipments. The shipment collection and distribution as a milk-run with different goods and a proportion of empty trips as well as other units than ton kilometers (such as volume, loading meters, number of pallets or containers) are considered.

In addition, the guideline describes the different scopes of emission and offers typical examples for mapping of processes and consumption. The definition of product carbon footprinting and the transfer to service activities such as transportation of goods are also given [4].

2.3 GLEC Framework

The voluntary guideline of the Global Emission Council (GLEC) offers also an approach for calculating transport emissions. In comparison with the DIN EN 16258 and the DSLV guideline the GLEC Framework has a global scope and aspires to be simple, transparent, accurate and flexible. The general approach of the framework is designed as follows:

- Plan: Define transport chains and Methodology
- Collect Data: Review data guidelines and identify gaps

- Calculate emissions: Find Emission factor and calculate for transport chain
- Define assumptions and follow the reporting instructions [1].

The defined transport chain can comprehend different modes of transports and includes transshipment processes, which can cause direct emission in Scope 1, or—if outsourced to a subcontractor—were added as Scope 3 emission. The consideration of transshipment centers with their direct fuel emissions as well as their indirect electricity emissions is included [1].

The GLEC Framework aims to be more precise in distance measurement. The actual distance, which is preferred in EN 16258 or Smartway, shall be used for Scope 1 emissions. The consumption factor is calculated by a division of fuel and ton kilometers, which is identical to other standards. For Scope 3 the use of planned distance with a correlation factor to allow deviation is preferred. This method refers to the difficulty that shipper often do not know the exact distance covered by a subcontractor. This is also a valid approach if actual distances aren't available.

In data collection the utilized fuel and electricity data are calculated, based on an average value for the provided service. This approach works for all different scopes, transport modes and transshipment activities.

In the Annex of the GLEC framework a large number of consumptions and emission factors are provided. Especially the consideration of different regions, different fuels and different vehicle types is taken into account and offers a lot of possible combinations to the user. Because of that an emission report given to a customer needs offer a lot of additional information explaining the utilized values to create a realistic picture of the logistics service. Only this additional information will allow the customer to choose a sustainable and green logistic service.

3 Methodology

First step in calculating carbon emissions of general good transports is the identification of the transport chain. As Fig. 2 shows the goods are picked up at different shippers and consolidated at the freight forwarders warehouse. Pre-carriage is usually operated by small trucks with a total weight of 7.5 or 12 t. The main carriage is conducted between the freight forwarder and the incoming forwarder by a heavy truck of 24t up to 40t total weight. After deconsolidating the main transport shipment into single shipments goods are forwarded to the consignee. On-carriage is produced by small trucks too, because usually collecting and delivering of goods is carried out on the same transport tour.

For calculating the carbon footprint emission of the transport chain, the following two formulas have to be utilized:

If the fuel consumption data of the transport, the route or the fleet are available:

$$Transport\,emission = \frac{Shipment\,weight\,in\,t}{Total\,weight\,in\,t} * Diesel\,consumption\,in\,l$$

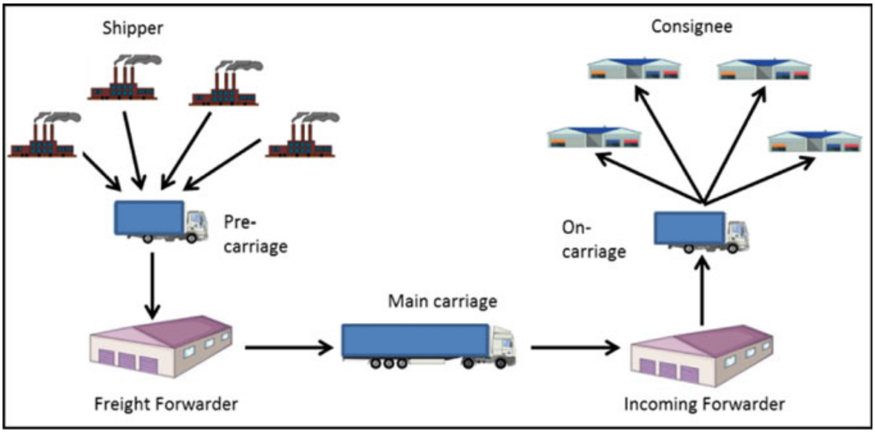

Fig. 2 General Cargo transport chain (own calculation)

$$* Emission\ factor\ in\ kg\ CO2e/tkm$$

In case of using the fourth approach of the EN 16258 or the GLEC Framework:

$$Transport\ Emission = Weight\ in\ t * Distance\ in\ km \\ * Emission\ faktor\ in\ kg\ CO2e/tkm$$

The weight is an easy available factor as it is given as shipment information of the shipper. Usually it is directly transferred into the TMS or noted on the shipment documents.

More difficult is the estimation of the distance as there are different ways of evaluation, which are also explained in the GLEC Framework. Following definitions of distances between two points are given [1]:

- Great circle distance: Direct connection, independent of mapping
- Shortest feasible distance: Theoretical distance, without consideration of weight and height restrictions or use of motorways and prevention city traffic
- Planned distance: Shortest distance, considering all operating conditions and trying to avoid delay due to heavy traffic
- Actual distance: Travelled distance, read by telematics e.g.

The first and the fourth option of distance determination are not in practical use. The great circle distance is unprecise and falsifies the result of an emission calculation. Especially for general cargo a direct connection is incorrect, because the pre-carriage and delivery are conducted as groupage trucking.

The actual distance is difficult to evaluate because parts of the transport chain or even the whole chain are operated by subcontractors. In many cases the logistics service provider only operates the transshipment facility. Running an own motor pool is often too expensive and needs too much administration and managing tasks.

Even if the logistics service provider operates own trucks, they usually only carry out parts of the transport routes and give the rest to subcontractors. For own trucks it is possible to calculate with actual distance, because transports are planned by own dispatchers supported by TMS and telematics. In contrast, subcontractors will plan the tour for deliveries and pick-ups by themselves with own telematics, tools and software solutions. Hence, actual distances won't be available for the logistics service provider. Even planned kilometers will only be available, if subcontractors work in the logistic provider's TMS or if they transfer their data into the TMS.

Planned distances will also cause problems in case routes have to be actualized due to new short-term pick-up stops, heavy traffic or scheduled services. In most cases logistics providers and freight forwarders will not apply this method as it is no valuable task. In most cases the logistics companies calculate the distance for carbon emissions by using a software solution, which puts out the shortest and most economical route regarding time and consumption. Famous solutions are map&guide, Google Maps and the distance calculation of emission reporting tools like EcoTransit [5]. By doing so, especially the calculation of mass data becomes much easier in handling. After weight and distance are evaluated and multiplied to the unit ton kilometers, an emission factor has to be chosen in order to transfer ton kilometer into carbon emission.

Regarding the four options of calculation mentioned by the EN 16258 only the use of values of an average own motor pool or standard factors from literature are feasible. If the different segments of the transport chain are operated by the own motor pool of the logistics service provider a specific factor can be calculated by dividing the used fuel in liters through the number of operated ton kilometers. The result can be multiplied with the fuel specific emission factor for the used fuel. In case of using subcontractors or non-available consumption data there are different factors given in GLEC framework and EN 16258. For that reason a reference is useful to be more comparable and to inform the customer about the methodology.

4 Findings and Comparison

The following chapter contains an example for the calculation of the mentioned methods in Chap. 3. At first, the different approaches of the EN 16258 with support of the DSLV guidelines will be calculated. At second, the same examples will be calculated with the restrictions of the GLEC Framework. In order to compare all results, the final unit after processing the calculations will be kilogram carbon dioxide equivalents per ton kilometer (kg CO2e/tkm). All Examples shall have a regard to "Well to Wheel" factors.

The example limits as follows:

Example Case:

- Shippers address (postal code and city): 49509 Recke
- Consignee address (postal code and city): 32657 Lemgo
- Shipment details: 1 pallet with 500 kg (120 × 80 × 150 cm).

Pre-carriage (PC):
Recke—Hub Osnabrueck (34 km):

- Roundtrip (98 km): 7.5t truck (2.3t loaded)
- Total Diesel consumption: 16.7 l.

Main Carriage (MC):
Hub Osnabrueck—Hub Bielefeld (78 km):

- Direct Trip: 40t truck (16t loaded)
- Total Diesel consumption: 24.3 l

On-carriage (OC):
HUB Bielefeld—Lemgo (28 km):

- Roundtrip (124 km): 7.5t truck (1.8t loaded)
- Total Diesel consumption: 15.8 l.

4.1 *Calculations with EN 16258 and DSLV Guidelines*

The EN 16258 offers four different options to calculate transport related emissions, which have been introduced in the second chapter and mentioned in Fig. 1.

- Emission factor Diesel: 3.24 kg CO2e/liter Diesel

Individual Measuring

The individual measuring is possible when all example limits are known:

$$
\begin{aligned}
Emission\ PC &= \frac{Shipment\ weight}{Total\ weight} * Diesel\ consumption \\
&\quad * Emission\ factor = \frac{0.5\mathrm{t}}{2.3\mathrm{t}} * 16.7\,\mathrm{l} * 3.24 \frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}} \\
&= 11.76\,\mathrm{kg\,CO_2e} \\
Emission\ MC &= \frac{Shipment\ weight}{Total\ weight} * Diesel\ consumption \\
&\quad * Emission\ factor = \frac{0.5\mathrm{t}}{16\mathrm{t}} * 24.3\,\mathrm{l} * 3.24 \frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}}
\end{aligned}
$$

$$
\begin{aligned}
&= 2.46\,\text{kg CO2e} \\
Emission\ OC &= \frac{Shipment\ weight}{Total\ weight} * Diesel\ consumption \\
&* Emission\ factor = \frac{0.5\text{t}}{1.8\text{t}} * 15.81 * 3.24 \frac{\text{kg CO2e}}{\text{l Diesel}} \\
&= 14.22\,\text{kg CO2e}
\end{aligned}
$$

The total emission for the individual measuring is the sum of the three different routes.

$$
\begin{aligned}
Total\ Emission &= Emission\ PC + Emission\ MC + Emission\ OC \\
&= 11.76\,\text{kg CO2e} + 2.46\,\text{kg CO2e} + 14.22\,\text{kg CO2e} \\
&= 28.44\,\text{kg CO2e}
\end{aligned}
$$

It is interesting to see, that the emission for the main transport account for only 8.6% of the total emission. Another fact is that the on-carrying transport needs less diesel but causes, due to the lower loading weight, more emission.

Measuring per route or type of vehicles

For the second calculation the limits of the example will be changed as the specific Diesel consumption is not known, but the average consumption of the operated vehicles is given:

- Average Diesel consumption 7.5t truck: 16.1 l/100 km
- Average Diesel consumption 40t truck: 31.7 l/100 km

With these assumptions the calculation is as follows:

$$
\begin{aligned}
Emission\ PC &= \frac{Shipment\ weight}{Total\ weight} * Distance \\
&* Average\ Diesel\ consumption * Emission\ factor \\
&= \frac{0.5\text{t}}{2.3\text{t}} * 98\,\text{km} * 16.1 \frac{\text{l Diesel}}{100\,\text{km}} * 3.24 \frac{\text{kg CO2e}}{\text{l Diesel}} \\
&= 11.11\text{kg CO2e}
\end{aligned}
$$

$$
\begin{aligned}
Emission\ MC &= \frac{Shipment\ weight}{Total\ weight} * Distance \\
&* Average\ Diesel\ consumption * Emission\ factor \\
&= \frac{0.5\text{t}}{16\text{t}} * 78\,\text{km} * 31.7 \frac{\text{l Diesel}}{100\,\text{km}} * 3.24 \frac{\text{kg CO2e}}{\text{l Diesel}} \\
&= 2.5\text{kg CO2e}
\end{aligned}
$$

$$
Emission\ OC = \frac{Shipment\ weight}{Total\ weight} * Distance
$$

$$
\begin{aligned}
&* \textit{Average Diesel consumption} * \textit{Emission factor} \\
&= \frac{0.5\mathrm{t}}{1.8\mathrm{t}} * 124\,\mathrm{km} * 16.1 \frac{\mathrm{l\,Diesel}}{100\,\mathrm{km}} * 3.24 \frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}} \\
&= 17.96\,\mathrm{kg\,CO2e}
\end{aligned}
$$

This will result in total emissions of:

$$
\begin{aligned}
\textit{Total Emission} &= \textit{Emission PC} + \textit{Emission MC} + \textit{Emission OC} \\
&= 11.11\,\mathrm{kg\,CO2e} + 2.5\,\mathrm{kg\,CO2e} + 17.96\,\mathrm{kg\,CO2e} = 31.56\,\mathrm{kg\,CO2e}
\end{aligned}
$$

The total emissions are a little bit higher than in the previous example. One reason for that is that effective driving during the on-carriage can't be taken into account when an average is taken.

Motor pool average

The averages of the different vehicles types aren't available in the third approach, but a general motor pool average of the freight forwarder is known:

- Average Diesel consumption Motor Pool: 25.7 l/100 km

With these assumptions the calculation is as follows:

$$
\begin{aligned}
\textit{Emission PC} &= \frac{\textit{Shipment weight}}{\textit{Total weight}} * \textit{Distance} \\
&\quad * \textit{Average Diesel consumption} * \textit{Emission factor} \\
&= \frac{0.5\mathrm{t}}{2.3\mathrm{t}} * 98\,\mathrm{km} * 25.7 \frac{\mathrm{l}\,\textit{Diesel}}{100\,\mathrm{km}} * 3.24 \frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}} \\
&= 17.74\,\mathrm{kg\,CO2e}
\end{aligned}
$$

$$
\begin{aligned}
\textit{Emission MC} &= \frac{\textit{Shipment weight}}{\textit{Total weight}} * \textit{Distance} \\
&\quad * \textit{Average Diesel consumption} * \textit{Emission factor} \\
&= \frac{0.5\mathrm{t}}{16\mathrm{t}} * 78\,\mathrm{km} * 25.7 \frac{\mathrm{l\,Diesel}}{100\,\mathrm{km}} * 3.24 \frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}} \\
&= 2.03\,\mathrm{kg\,CO2e}
\end{aligned}
$$

$$
\begin{aligned}
\textit{Emission OC} &= \frac{\textit{Shipment weight}}{\textit{Total weight}} * \textit{Distance} \\
&\quad * \textit{Average Diesel consumption} * \textit{Emission factor} \\
&= \frac{0.5\mathrm{t}}{1.8\mathrm{t}} * 124\,\mathrm{km} * 25.7 \frac{\mathrm{l\,Diesel}}{100\,\mathrm{km}} * 3.24 \frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}} \\
&= 28.68\,\mathrm{kg\,CO2e}
\end{aligned}
$$

This will result in total emissions of:

$$
\begin{aligned}
Total\ Emission &= Emission\ PC + Emission\ MC + Emission\ OC \\
&= 17.74\,\text{kg CO2e} + 2.03\text{kg CO2e} + 28.68\,\text{kg CO2e} \\
&= 48.45\,\text{kg CO2e}
\end{aligned}
$$

The total emissions calculated by using the third approach are significant higher than in the previous examples. Especially the on-carrying has been increased, due to the high average consumption values of the motor pool.

Calculating with standard emission factors

The fourth approach is based on the ton kilometers. Only distances between shipper and Hub, Hub and Hub and between Hub and consignee as well as the weight of the shipment are known. The standard emissions factors are as follows:

- Emission factor 7.5t: 0.192 kg CO2e/tkm
- Emission factor 40t: 0.072 kg CO2e/tkm

With these parameters the calculation will be done as follows:

$$
\begin{aligned}
Emission\ PC &= Shipment\ weight * Distance * Emission\ factor \\
&= 0.5\text{t} * 34\,\text{km} * 0.192\,\text{kg CO2e} = 3.26\,\text{kg CO2e} \\
Emission\ MC &= Shipment\ weight * Distance * Emission\ factor \\
&= 0.5\text{t} * 78\,\text{km} * 0.072\,\text{kg CO2e} = 2.81\,\text{kg CO2e} \\
Emission\ PC &= Shipment\ weight * Distance * Emission\ factor \\
&= 0.5\text{t} * 28\,\text{km} * 0.192\text{kg CO2e} = 2.67\,\text{kg CO2e}
\end{aligned}
$$

This will result in total emissions of:

$$
\begin{aligned}
Total\ Emission &= Emission\ PC + Emission\ MC + Emission\ OC \\
&= 3.26\,\text{kg CO2e} + 2.81\,\text{kg CO2e} + 2.67\,\text{kg CO2e} \\
&= 8.74\,\text{kg CO2e}
\end{aligned}
$$

The total emissions of the forth approach are by far the lowest in comparison with all other approaches, even if no specific data is available.

4.2 Calculation with GLEC Framework

For the GLEC framework two different calculation models will be analyzed. At first, with the real measured consumption and at second with standard emission factors [1].

Calculation based on real measured data

The scope 1 emission of transports and the scope 1 and 2 emissions for transshipments or cargo handling are in focus of the GLEC approach. At first, scope 1 emissions are taking the real fuel consumption of trucks into account.

$$\begin{aligned} Fuel\,used &= Fuel\ PC + Fuel\ MC + Fuel\ OC = 16.7\,\mathrm{l} + 24.3\,\mathrm{l} + 15.8\,\mathrm{l} \\ &= 56.8\,\mathrm{l\,Diesel} \end{aligned}$$

At second, an input factor regarding the performance is needed, which shall be indicated as ton-kilometers (tkm), which means that the weight and the distance have to be multiplied. As the distance is only given as planned kilometers a deviation factor of 10% shall be considered in the calculation [1].

$$\begin{aligned} Performance &= tkm\ PC + tkm\ MC + tkm\ OC \\ &= 2.3\mathrm{t} * 98\,\mathrm{km} * 1.1 + 16\mathrm{t} * 78\,\mathrm{km} * 1.1 + 1.8\mathrm{t} * 124\,\mathrm{km} * 1.1 \\ &= 247.9\,\mathrm{t\,km} + 1372.8\,\mathrm{t\,km} + 245.5\,\mathrm{t\,km} = 1866.2\,\mathrm{t\,km} \end{aligned}$$

The emission factor, which results from the above mentioned analysis is as follows:

$$\text{Emission factor Transport} = \frac{\text{Fuel used}}{\text{Performance}} = \frac{56.8\mathrm{l}\ Diesel}{1866.2\ t\mathrm{km}} = 0.0304\,\mathrm{l\,Diesel/\,t\,km}$$

For the single shipment of one pallet with 500 kg the transport emission of carbon dioxide equivalents are calculated as follows. The conversion Factor for Diesel as per GLEC framework is defined as 3.9 kg CO2e for Europe [1].

$$\begin{aligned} Transport\ emission &= tkm * Emission\ factor\ Transport * Conversion\ factor \\ &= (0.5\mathrm{t} * 34\,\mathrm{km} * 1.1 + 0.5\mathrm{t} * 78\,\mathrm{km} * 1.1 + 0.5\mathrm{t} * 28\,\mathrm{km} * 1.1) \\ &\quad * 0.0304\frac{\mathrm{l\,Diesel}}{\mathrm{t\,km}} * 3.9\frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}} = 77\ t\mathrm{km} * 0.0304\frac{\mathrm{l\,Diesel}}{\mathrm{t\,km}} \\ &\quad * 3.9\frac{\mathrm{kg\,CO2e}}{\mathrm{l\,Diesel}} = 9.129\,\mathrm{kg\,CO2e} \end{aligned}$$

After calculating the transport emission, transshipment emissions are the second relevant part for the calculation of the total emission and have to be added. For this calculation the shipment weight is needed as well as an emission factor based on the consumed energy and the performance.

$$Emission\ factor\ Transshipment = \frac{(Energy\ in\ kWh + Heating\ in\ kWh)}{Tonnes\ of\ outgoing\ freight}$$

The following example calculates the emission factors of the Hub Osnabrueck (OSN), based on the energy consumption per day. Furthermore the same emission factors have to be calculated for Bielefeld (BFE), the second hub in the given example.

$$Emission\ factor\ OSN = \frac{(4046.7\,\mathrm{kWh} + 2332.9\,\mathrm{kWh})}{6655.9\mathrm{t}} = 0.958\frac{\mathrm{kWh}}{\mathrm{t}}$$

$$Emission\ factor\ BFE = \frac{(1056.4\,\mathrm{kWh} + 438.2\,\mathrm{kWh})}{1887.8\mathrm{t}} = 0.791\frac{\mathrm{kWh}}{\mathrm{t}}$$

The transshipment emission is calculated as follows with consideration of a conversation factor for energy of 0.622 kg CO2e per kWh [1].

$$\begin{aligned} Transshipment\ Emission &= Shipment\ weight * (Emission\ factor\ OSN \\ &\quad + Emission\ Factor\ BFE) \quad * Conversion\ factor \\ &= 0.5\mathrm{t} * 1.749\frac{\mathrm{kWh}}{\mathrm{t}} * 0.622\frac{\mathrm{kgCO2e}}{\mathrm{kWh}} \\ &= 0.543\,\mathrm{kg\,CO2e} \end{aligned}$$

These two parts, the transportation and the transshipment emissions will be added in order to create a result for the total emission based on the GLEC Framework:

$$\begin{aligned} Total\ Emission &= Transport\ Emission + Transshipment\ Emission \\ &= 9.129\,\mathrm{kg\,CO2e} + 0.543\,\mathrm{kg\,CO2e} = 9.673\,\mathrm{kg\,CO2e} \end{aligned}$$

Calculation based on standard emission factors

The calculation based on the standard emission factors is quite simple because only the emission factor is changing—from a specific and measured value to a default value. The value of ton kilometers (tkm) and the transshipment activities and emission will remain the same. The standard emission factors for the used vehicles as per GLEC framework:

- Emission factor 7.5t: 0.193 l Diesel/tkm
- Emission factor 40t: 0.072 l Diesel/tkm

The edited formula for transport emission as follows:

$$\begin{aligned} Transport\ emission &= \left(\sum tkm * Emission\ factor\ Transport\right) \\ &\quad * Conversion\ factor \\ &= \left(0.5\mathrm{t} * 34\,\mathrm{km} * 1.1 * 0.193\frac{\mathrm{l\,Diesel}}{\mathrm{t\,km}} + 0.5\mathrm{t} * 78\,\mathrm{km} * 1.1\right. \\ &\quad \left. * 0.023\frac{\mathrm{l\,Diesel}}{\mathrm{t\,km}} + 0.5\mathrm{t} * 28\,\mathrm{km} * 1.1 * 0.193\frac{\mathrm{l\,Diesel}}{\mathrm{t\,km}}\right) \end{aligned}$$

$$* \, 3.9 \frac{\text{kg CO2e}}{\text{l Diesel}} = (3.61\,\text{l Diesel} + 0.987l\,\text{Diesel} + 2.97l\,\text{Diesel})$$
$$* \, 3.9 \frac{\text{kg CO2e}}{\text{l Diesel}} = 29.51\,\text{kg CO2e}$$

As mentioned above, the emission for the transshipment will remain the same, to that the total emission will be calculated as follows:

$$\begin{aligned} Total\ Emission &= Transport\ Emission + Transshipment\ Emission \\ &= 29.51\,\text{kg CO2e} + 0.543\,\text{kg CO2e} = 30.053\,\text{kg CO2e} \end{aligned}$$

5 Conclusion

The calculation of carbon emissions, as it has been conducted in a given example in the previous chapter, offers a lot of different approaches to provide carbon emission and carbon foot-printing. Logistics services turn out to be difficult, due to the fact that varying information for calculation is given. In case of general cargo it is even more complicated as the transport chain is composed of multiple single transports as well as transshipment handling, which is a major characteristic of general cargo logistics operations. Especially when the transport chain includes subcontractors or partners, the data availability and quality decreases and the logistics service provider has to deal with assumptions and average or standard values and factors. This leads to miscellaneous and imprecise results as well as different footprint calculation results of logistics services.

At the same time, both standards which have been compared in this paper have some similarities regarding their approaches and definitions. For instance, both standard emission factors are comparable. Furthermore, both standards demand a lot of knowledge and information from the logistics companies.

Clarifying all requirements of the different standards and the calculation methods, with various parameters and varying emission factors as well as deviation factors or conversion factors, is a huge challenge. Especially when the scientific approach of the standards and guide-lines contain complex and confusing paragraphs. Smaller logistics service providers and freight forwarders don't have the manpower and the knowledge to decide which standard or which procedure is most valid or fits best to the practice. The difference in the results of the calculations also proof a deviation between the standards and approaches, but it is unclear what the correct result is or which one is more valid.

As emission reports are requested by the customer, the logistics service providers and freight forwarders need to deliver correct and valid data. The customers want to use the calculated emission data of the logistics services for their own sustainability reporting and for the eco-logical footprint of products. Both parties need specific data as environmental aspects such as carbon emission are getting into the focus of society.

Our recommendation, based on the comparison of both mentioned existing standards, is a combination of elements out of both approaches. The approach of segmentation of the transport chain as utilized within the GLEC framework is feasible. In combination with the use of average values of fuel consumption in regard to the vehicle class (not the entire motor pool) and the existing method to calculate transshipment emissions according to the GLEC framework, a functional and at the same time valid methodology is possible. In addition a set of consistent default values is needed. A methodology like this, combining the advantages of both describes calculation standards—being as detailed as needed and as practical as possible, has the potential to become a global standard.

When carbon emissions of logistics become an evaluation criterion for logistics services and the performance of logistics companies or even a decision criterion for tendering or business connections, a clear and explicit approach for emission calculation is needed. It has to be comparable in the calculation steps as well as in the result. The method has to be easy understandable for all parts of the logistics and the supply chain, so that the participants can check and review the data of the other participants. By doing so, the support of smaller or underdeveloped partners would be easier, too. The consumer in particular and the society as a whole wants to have better conditions to rate and rank their products based on the footprint and wants to have a certain option to buy the product with the smallest footprint.

As logistic chains becoming more and more complex, a worldwide obligatory standard needs to be established, in order to enable comparability of carbon footprint calculations under same conditions and with the same understanding. This will ensure fair and equal competition.

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