Sustainable Production, Life Cycle Engineering and Management

Series Editors: Christoph Herrmann, Sami Kara

Franziska Hesser · Iris Kral · Gudrun Obersteiner · Stefan Hörtenhuber · Martin Kühmaier · Vanessa Zeller · Liselotte Schebek *Editors*

Progress in Life Cycle Assessment 2021

MOREMEDIA

Springer

Sustainable Production, Life Cycle Engineering and Management

Series Editors

Christoph Herrmann, Braunschweig, Germany

Sami Kara, Sydney, Australia

SPLCEM publishes authored conference proceedings, contributed volumes and authored monographs that present cutting-edge research information as well as new perspectives on classical fields, while maintaining Springer's high standards of excellence, the content is peer reviewed. This series focuses on the issues and latest developments towards sustainability in production based on life cycle thinking. 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.

Indexed in Scopus

To submit a proposal or request further information, please use the PDF Proposal Form or contact directly: Petra Jantzen, Applied Sciences Editorial, email: petra.jantzen@springer.com

Franziska Hesser · Iris Kral · Gudrun Obersteiner ·
Stefan Hörtenhuber · Martin Kühmaier ·
Vanessa Zeller · Liselotte Schebek
Editors

Progress in Life Cycle Assessment 2021

Springer

Editors
Franziska Hesser
Wood K Plus—Competence Centre for Wood Composites and Wood Chemistry, Kompetenzzentrum Holz GmbH
Linz, Austria

Gudrun Obersteiner
Department of Water, Atmosphere and Environment
University of Natural Resources and Life Sciences Vienna
Vienna, Austria

Martin Kühmaier
Department of Forest- and Soil Sciences
University of Natural Resources and Life Sciences Vienna
Vienna, Austria

Liselotte Schebek
Technical University of Darmstadt
Material Flow Management and Resource Economy
Darmstadt, Germany

Iris Kral
Department of Sustainable Agricultural Systems
University of Natural Resources and Life Sciences Vienna
Vienna, Austria

Stefan Hörtenhuber
Department of Sustainable Agricultural Systems
University of Natural Resources and Life Sciences Vienna
Vienna, Austria

Vanessa Zeller
Technical University of Darmstadt
Material Flow Management and Resource Economy
Darmstadt, Germany

ISSN 2194-0541 ISSN 2194-055X (electronic)
Sustainable Production, Life Cycle Engineering and Management
ISBN 978-3-031-29293-4 ISBN 978-3-031-29294-1 (eBook)
https://doi.org/10.1007/978-3-031-29294-1

This Springer imprint is published by the registered company Springer Nature Switzerland AG
The registered company address is: Gewerbestrasse 11, 6330 Cham, Switzerland

Contents

1 **Resources in a Circular Economy with a Focus on Land Use** 1
Franziska Hesser and Theresa Krexner

2 **The Climate Impact of the Usage of Headphones and Headsets** 7
Tayla Herrmann, Anna Zimmerer, Claus Lang-Koetz, and Jörg Woidasky

3 **Implementing Ecodesign During Product Development: An Ex-Ante Life Cycle Assessment of Wood-Plastic Composites** 23
Nadine Brunnhuber, Andreas Windsperger, Enrique Alejandro Perdomo Echenique, and Franziska Hesser

4 **Background Data Modification in Prospective Life Cycle Assessment and Its Effects on Climate Change and Land Use in the Impact Assessment of Artificial Photosynthesis** 41
Lukas Lazar and Andreas Patyk

5 **Understanding Soil Organic Carbon Dynamics of Short Rotation Plantations After Land Use Change—From Establishment to Recultivation** 65
Enrique Alejandro Perdomo Echenique and Franziska Hesser

6 **Assessment of Greenhouse Gas Emission Reduction from Biogas Supply Chains in Germany in Context of a Newly Implemented Sustainability Certification** 85
Nora Lange, David Moosmann, Stefan Majer, Kathleen Meisel, Katja Oehmichen, Stefan Rauh, and Daniela Thrän

7 Proposing a Multi-level Assessment Framework for Social LCA and Its Contribution to the Sustainable Development Goals . . . 103
Daniela Groiß-Fürtner, Claudia Mair-Bauernfeind, and Franziska Hesser

8 Life Cycle Sustainability Assessment for Sustainable Bioeconomy, Societal-Ecological Transformation and Beyond 131
Walther Zeug, Alberto Bezama, and Daniela Thrän

Chapter 1
Resources in a Circular Economy with a Focus on Land Use

Franziska Hesser and Theresa Krexner

Abstract This preface introduces the setting of the 16th Ökobilanzwerkstatt, a forum for early-career life cycle assessment (LCA) researchers. Under the motto "resources in a circular economy with a focus on land use," presentations of ongoing research were given from September 22nd to 24th 2021. Highlights compiled in this book cover case studies for circular economy strategies and emerging technologies in the field of bioeconomy, the introduction of conceptual frameworks especially related to social LCA, and application of new modeling approaches with a focus on energy provision and land use. The compilation of topics, scientific contributions, and practical contexts mirrors the interdisciplinarity of LCA research and provides a snapshot of the breadth, such as depth of progress in LCA theory and application.

Keywords Life cycle assessment · Ökobilanzwerkstatt · Circular bioeconomy · Social LCA · Land use · Sustainability

1.1 Contextualization

Against the background of climate change, bio-based resources are being utilized at an accelerating rate, and land use and changes to how land is used are politically justified. In the European context, the bioeconomy concept has been developed as policy instrument embedded in larger economic growth strategy, and it is based on the substitution of fossil resources for the development of biotechnology as a new economic sector (see Birner 2018 for elaboration on bioeconomy concepts). Birner (2018) also mentions fundamental critique which addresses the contradiction of the

F. Hesser (✉)
Wood K plus—Competence Centre for Wood Composites and Wood Chemistry, Kompetenzzentrum Holz GmbH, Altenberger Straße 69, 4040 Linz, Austria
e-mail: f.hesser@wood-kplus.at

T. Krexner
Department of Sustainable Agricultural Systems, Institute of Agricultural Engineering, University of Natural Resources and Life Sciences, Peter-Jordan-Straße 92, 1190 Vienna, Austria

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*, Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_1

economic growth paradigm within planetary boundaries and so called "greenwashing critique" which raises awareness that use of bio-based resources does not contribute to environmental protection and sustainable development per se (Birner 2018)—which makes the transparent assessment of such even of greater importance.

For the transition to a sustainable circular bioeconomy and toward the overarching ambition of climate protection, a whole set of strategic policy agendas frame the societal and economic environment. For example, the United Nations' Sustainable Development Goals (United Nations 2015) and the Paris Climate Agreement (European Commission s.a.) represent global, overarching societal objectives, while the Circular Economy Action Plan 2020 (European Commission 2020), one of the main parts of the European Green Deal (European Commission 2021), aims to pave the way for a digital and green transition to a circular economy.

Aiming at the aforementioned, the sustainability assessment of human induced activities manifested in products, processes, and services seems to be a necessity, but is in fact not performed on a wide scale. By combining the three methods, environmental life cycle assessment (LCA), social LCA (sLCA), and life cycle costing (LCC), we can perform a comprehensive life cycle sustainability assessment. These methods generate system understanding and sustainability knowledge to lever improvement potentials towards societally desired goals. Although environmental LCA is the most advanced method for quantitative assessment of potential environmental impacts, the changing societal challenges and requirements reveal several limitations within these methods, which reflects in the continuous advancement, specialization, and diversification of sustainability assessment.

1.2 The 16th Ökobilanzwerkstatt

Increasing requirements for impact assessment are also reflected in academia, where new applications for LCA, new modeling approaches, new inventory tools and method combinations for the interpretation, etc. are developed. Early-career researchers of various disciplines and research contexts characterize an important pillar of the community that argues the necessity to provide a forum for scientific communication and personal exchange. The Ökobilanzwerkstatt, established by Professor Liselotte Schebek from Technical University Darmstadt in Germany in 2005, is such a forum and has since then—except 2020—supported an early-career LCA researchers' conference yearly with different hosts across mainly German universities. From September 22nd to 24th 2021, Wood K plus organized the 16th Ökobilanzwerkstatt at the University of Natural Resources and Life Sciences, Vienna (BOKU). Due to the Corona pandemic and the governmental measures, the conference planned for 2020 was postponed to 2021 and finally had to be realized in an online setting.

Forty international students from Austrian and German universities participated under the motto "resources in a circular economy with a focus on land use." The presentations were grouped in eight thematic sessions chaired by postdocs from Wood K plus, BOKU, and TU Darmstadt and framed by two keynote speeches.

The first one was held by Cécile Bessou, the research director at the Agricultural Research Center for Development (CIRAD), who gave a presentation on ‘Optimizing trade-off between feasibility and accuracy in carbon and nitrogen modelling within LCA.” She highlighted the importance of understanding and comprising interconnections of different nutrient cycles for the LCA interpretation.

The second keynote speaker Johannes Lindorfer from the Energy Institute of the Johannes Kepler University in Linz, Austria, who presented the “Technology Collaboration Program by the International Energy Agency Bioenergy Task 42—Biorefining in a Circular Economy.” He highlighted the developed method technology, economic, environmental (TEE) assessment of novel biorefinery technology pathways.

To enable networking despite the conference being held online, a broad social program was offered. In virtual events the participants had the chance to go on a virtual tour through BOKUs’ university campus, to meet the core group of the BOKU-LCA Platform, and to get to know the Austrian LCA community.

1.2.1 Wood K Plus

Wood K plus is a research organization in the area of wood and wood-related renewable resources in Europe and was initially developed about 20 years ago as research project driven by BOKU professors from the department for social and economic sciences. Wood K plus runs a Competence Center for Excellent Technologies (COMET K1) which focusses on “WOOD: Transition to a sustainable bioeconomy,” COMET is a long-term R&D funding program for science and industry to support application-oriented cutting-edge research in Austria.

The core competences of Wood K plus are materials research and process technology along the complete value chain, from raw material to finished products. Wood K plus develops methods and basic materials and performs applied research on the economy-science interface in order to enable resource-efficient management in the circular bioeconomy. The team “Sustainable Innovation and Impact Assessment” (formerly known as team “Market Analysis and Innovation Research”) conducts research at the intersection of technology and society with the aim of successfully shaping innovation processes by providing information on ecological, economic, and social questions on the market. The team is pleased to co-operate with the BOKU-LCA Platform on LCA research activities and honored to organize the 16th Ökobilanzwerkstatt in 2021 at BOKU.

1.2.2 BOKU-LCA Platform

The LCA method has already been used at BOKU for more than 15 years, but the BOKU-LCA Platform was just founded in 2013 after three researchers from different BOKU Departments became acquainted at a SETAC conference and then decided to collaborate. The overall aim is to build a network of LCA practitioners (from beginners to experts) at BOKU and provide a platform to exchange experience since the method is used in various research fields. An important element of this is the possibility to discuss methodological aspects in a constructive and competent group and to further develop the method based on BOKU-specific backgrounds, e.g., linking production and recycling or energy and nutrient balances. This helps to solve problems faster and more effectively, contributes to knowledge transfer, and helps to keep the professional standard of LCA at a high level. In addition, the broad LCA-knowledge and practical experience of the members is provided in lectures, but also the best possible support can be offered in the supervision of master's theses and dissertations engaging in LCA topics.

Further, the BOKU-LCA platform wants to position itself as main point of contact regarding the topic of quantitative sustainability assessment not only at BOKU, but also for external inquiries, which makes it indispensable to raising the visibility of LCA to external parties as well.

The agendas of the BOKU-LCA platform are managed by a steering group, which is composed of one representative from each of five different departments (Civil Engineering and Natural Hazards, Sustainable Agricultural Systems, Forest and Soil Sciences, Water-Atmosphere-Environment, and Economics and Social Sciences), who meet regularly for exchange and presenting the latest research results. Since 2019, the Centre for Bioeconomy supports these cross-departmental projects through coordination and communication within BOKU but also with relevant stakeholders in society, with the goal of further developing quantitative applications of potential environmental impact assessments through BOKU-specific workshops and presentations.

Currently, a wide range of topics are assessed by BOKU-LCA platform members, such as assessment and certification of buildings, sustainability analysis of food chains, assessment of agricultural systems, supply chains of wood, assessment and improvement in terms of environmental impacts of new materials, ecological effects of waste prevention measures in various fields, and assessment of sustainable bioenergy.

Due to the growing importance of sustainability assessment of human-induced activities for decision-makers, consumers, and companies in the last few years, the BOKU-LCA platform is taking on an increasingly important role as a nucleus of expertise. Further, as more requirements through standardization and guidelines increase the complexity of the LCA, a common, cross-institutional, collaborative approach will be beneficial for all researchers working in the LCA field.

1.3 The Book Progress in Life Cycle Assessment 2021

The presentations and discussions at the 16th Ökobilanzwerkstatt 2021 led to the following conclusions:

- Prospective LCA faces the problem of producing "more unknowns";
- Different approaches to LCA are helpful for generating different insights and handling data-related constraints, such as data requirements, availability, quality, accessibility, and confidentiality;
- The assessment of land use and land use change is connected to resource allocation and implications of possible societal conflicts need to be addressed in social LCA;
- Currently, most LCAs are centered around the global warming potential, leading to blind spots on other potential environmental impacts;
- Perceiving a shortcoming in single indicator assessments and asking for systemic analyses, the students call for future progress in consequential LCA.

In this realm, this book is compiled of articles written by participants of the 16th Ökobilanzwerkstatt 2021 based on their conference presentations. The contributions reach from master's degree studies to early and more advanced doctoral research. As most of the projects out of which the presented LCA studies were drawn are still ongoing, the articles give previews beyond the state of the art. Articles address a wide range of topics, such as case studies of promising strategies in the context of circular economy and new bioeconomy technologies, design of conceptual frameworks, especially related to social LCA, and the use of novel modeling approaches with a focus on energy supply and land use. The diversity of topics covered, scientific contributions, and practical contexts reflect the policy context, but especially the interdisciplinarity and also transdisciplinarity within LCA research, and provide a snapshot of the breadth and depth of research currently being conducted by early-career LCA researchers in terms of method and application.

Acknowledgements This work received funding by the Bio Based Industries Joint Undertaking under the European Union's Horizon 2020 research and innovation programme under grant agreement number 745874, the Austria Research Promotion Agency (FFG) under the COMET program grant number 865905 and the Technology Collaboration Programme by IEA Bioenergy Task 42: Biorefining in a Circular Economy.

References

Birner R (2018) Bioeconomy concepts. In: Lewandowski I (eds) Bioeconomy. Springer, Cham. https://doi.org/10.1007/978-3-319-68152-8_3

European Commission (s.a.) Climate Action Paris Agreement. https://ec.europa.eu/clima/eu-action/international-action-climate-change/climate-negotiations/paris-agreement_en. Last accessed 22/08/2022

European Commission (2020) Environment-Circular economy action plan. https://ec.europa.eu/environment/strategy/circular-economy-action-plan_en. Last accessed 22/08/2022

European Commission (2021) A European Green Deal. Striving to be the first climate-neutral continent. https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en. Last accessed 22/08/2022

United Nations (2015) Transforming our world: the 2030 agenda for sustainable development. https://sustainabledevelopment.un.org/content/documents/21252030%20Agenda%20for%20Sustainable%20Development%20web.pdf. Last accessed 22/08/2022

Chapter 2
The Climate Impact of the Usage of Headphones and Headsets

Tayla Herrmann, Anna Zimmerer, Claus Lang-Koetz, and Jörg Woidasky

Abstract Based on disassembly studies, a life cycle assessment of the climate impact of the wireless over-ear headphone model Jabra Evolve2 85 (without charging station) is conducted regarding the life cycle phases of manufacturing, packaging, distribution, use and disposal. The total weight of all components is 280.7 g. The materials can be categorized into polymers (61.7%), metals (20.9%), circuit boards (4.8%), Li-ion battery (4.6%), foam (3.5%), cables (3.0%) and unidentifiable polymers (1.7%). The functional unit is defined as the wireless audio transmission through a stereo headphone over its lifetime. The lifecycle assessment results in a global warming potential of 12.17 kg CO_2-Eq with a contribution of the manufacturing phase of 81.2%, based on an assumed lifetime of 2,600 using hours. In the context of a sensitivity analysis, a repair scenario of a battery replacement of the over-ear headset is modelled. Assuming a doubled lifetime, the global warming potential per hour is reduced from 4.7 g CO_2-Eq/h to 2.4 g CO_2-Eq/h.

Keywords LCA · Headphones · Dismantling · Life cycle data inventory

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/978-3-031-29294-1_2.

T. Herrmann (✉) · A. Zimmerer · C. Lang-Koetz · J. Woidasky
Institute for Industrial Ecology, Pforzheim University, Tiefenbronner Str. 65, 75175 Pforzheim, Germany
e-mail: herrmany@hs-pforzheim.de

J. Woidasky
Institute for Materials and Material Technologies, Pforzheim University, Tiefenbronner Str. 65, 75175 Pforzheim, Germany

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*,
Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_2

2.1 Introduction

Headphones are electronic devices worn on the head to transmit sound to human ears both in the business world and for private usage. They facilitate peoples' lives as they, for example, allow having calls or listening to music anywhere at any time without disturbing the environment. The usage of headphones is increasing: In 2020, around 15.5 million headphones were sold in Germany, an increase of 6.8% compared to 2019 (gfu Consumer & Home Electronics GmbH and Growth from Knowledge 2020).

GN Store Nord A/S is an international company based in Denmark which provides hearing, audio, and collaboration solutions under the brand name "Jabra". Their sustainability targets include goals referring to the sustainability of their products, packaging, manufacturing, and distribution (GN Audio A/S 2020a). In 2020, the trade-in-system "Jabra Green Initiative – Recycle and Benefit" started in collaboration with the German company TechProtect GmbH (GN Audio A/S n.d.a), a company that offers marketing integrated services and take back solutions. In this context, Pforzheim University students analyzed the environmental impact of one Jabra headphone model in cooperation with these two companies. The goal of this work was to identify the components and materials of this particular headphone model, the Jabra Evolve2 85 (28599-989-999), and to assess its climate impact measured as contribution to the global warming potential. To achieve this goal, dismantling trials were conducted, followed by a life cycle assessment including a sensitivity analysis with a repair scenario.

2.2 Background

2.2.1 Headphones' Components and Materials

The main elements of cordless headphones are the ear pads, ear cases, main circuit boards, speakers, the microphone arm, the headband, and the battery, displayed in Fig. 2.1.

The product data sheet of the Jabra Evolve2 85 (GN Audio A/S 2020b) and ten additional data sheets of selected current cordless on-ear and over-ear headphone models were identified in an online search. In order to provide some benchmark information and to verify the Jabra Evolve2 85 as a representative product, a comparison was conducted. For this purpose, all available information of the products' data sheets covering type of headphone, weight, components, materials, battery, microphones, and speakers were compiled. The products' data sheets of the following comparable headphone models were considered: Jabra Evolve 65 (GN Audio A/S 2020c), Jabra Elite 85 h (GN Audio A/S 2019), Poly Voyager 4200 Office & UC (Plantronics Inc. 2021), Logitech Zone Wireless (Logitech n.d.), JBL Live650BTNC (Harman International Industries Inc. 2019), Sony WH-1000XM3 (Sony Europe

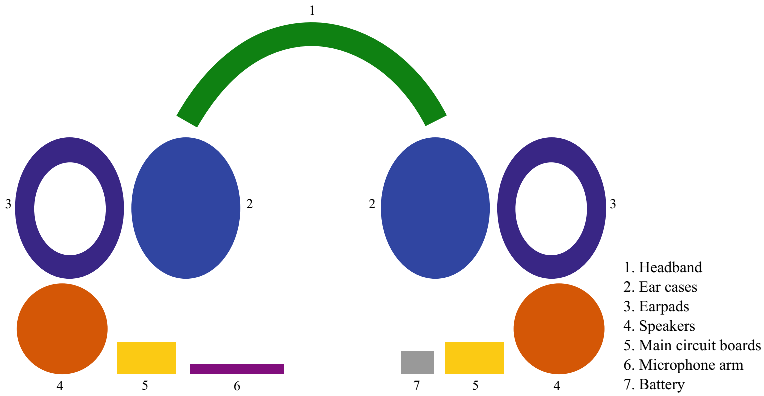


Fig. 2.1 Main components of a cordless headphone (schematic)

B.V. n.d.), ATH-M50xBT (audio-technica n.d.), Teufel Supreme on (Lautsprecher Teufel GmbH n.d.), B&O Beoplay H9 3rd gen (Bang & Olufsen n.d.), Panasonic RP-HD610N (Panasonic Deutschland n.d.).

The range of the stated weight is between 150 and 310 g. The stated weight of eight products is higher than 250 g, including the Jabra Evolve2 85 with 286 g. In the selected products, the loudspeaker housing consists of stainless steel, aluminum, and polymers. The headband is often covered in foam and wrapped with fabric or synthetic leather. The same materials can be found in the ear cushions. If specified, the battery is a lithium-ion-battery or a lithium-polymer-battery. This applies for the assessed product and seven additional products. Three different microphone types can be found: microelectromechanical systems (MEMS), electret condenser microphones (ECM), and condenser microphones. The assessed headphone contains 4 analogue MEMS (microelectromechanical systems) microphones and 6 digital MEMS microphones. In at least five of the eleven headphones, electrodynamic speakers are installed. In the other products' data sheets, the type of speakers is not specified, including the Jabra Evolve2 85. The speaker's diameter is stated in all product data sheets and ranges between 28 and 45 mm. The speakers' diameter of the assessed product is 40 mm.

2.2.2 Expected Lifetime of Components and Headphone

The lifetime of rechargeable lithium-ion-batteries can be measured and stated in two ways:

1. calendrical lifetime (time, for example in years, until the battery has a certain capacity left, e.g. 80% of the nominal capacity) and

2. cycle lifetime (number of charging cycles until the battery has a certain capacity left, e.g. 80% of the nominal capacity) (Job 2020).

An average battery provides approximately 500–1000 charging cycles until it loses 20% of its capacity (Korthauer 2013). In simulations by Maia et al., approximately 400 cycles with optimized charging are observed until a capacity of 80% is reached (Maia et al. 2019). Based on an assumption of a cycle lifetime of 500 cycles and recharging every third day, this would correspond to a calendrical lifetime of 4.1 years. This is in line with other literature (Broussely et al. 2001). According to Schulze and Buchert, a lifetime of 8 years with a standard deviation of 2 years can be assumed for acoustic transducers with neodymium-iron-boron magnets (Schulze and Buchert 2016). The lifetime of the ear pads is depending on material and usage, but the component is considered as rather susceptible to wear due to its position, purpose, and the large offer for spare parts, for example by Jabra (GN Audio A/S n.d.b). In a Swiss case study regarding the service lifetime, storage lifetime, and disposal pathways of electronic equipment, statistical data for several electronic devices, including headsets, were collected. The average service lifetime of already disposed or stored headsets is found to be at 3.3 years (Thiébaud-Müller et al. 2018). In agreement with the producer, a value of 2 years was applied in the life cycle assessment of the model Jabra Evolve2 85, based on the product's warranty time period. According to the manufacturer, a 2,600 using hour lifetime can then be assumed. This value can be justified by the assumption of a product use intensity of five hours per day, five days per week, and the product's warranty time period of two years.

2.3 Methods

Dismantling trials were carried out in the laboratory for sustainable product development at Pforzheim University as a part of a student project in the period from 31/03/2021 to 21/04/2021 by the authors. In total, three headphones Jabra Evolve2 85 (Product No. 28599-989-999) were dismantled which had been supplied by Jabra from product returns. With the first product, the construction, the components, and an appropriate way for dismantling were identified. One of the remaining products was used for the documentation of the single dismantling steps to homogenous material, including the needed time and tools. Conventional workshop tools only were used for this work, listed in Table 2.1.

The dismantling of the third headphone served for the documentation of the components and the development of the bill of materials. For this, the precision scale Kern 573–46 was used. Manufacturer's marks on the polymer parts, X-ray fluorescence analysis (XRF, Thermo Fisher Scientific, Niton XL2 air 980), and attenuated total reflection (ATR, Bruker, Alpha Platinum) analysis were used for material identification. For the ATR analysis, the BPAD.S01 (Brucker Optics ATR-Polymer Library) and Demolib.s01 (General Library IR) data bases were used. All measurements were carried out threefold.

Table 2.1 Tools used for dismantling

No	Tools
1	Screwdriver, crosstip, C.K. precision, N0.0
2	Screwdriver, C.K precision, T5
3	Screwdriver, C.K precision, T6
4	Utility knife, KS Tools
5	Cross-cut chisel, KS Tools, 5 mm
6	Flat chisel, KS Tools, 12 mm
7	Hammer, Projahn, DIN 1041 500
8	Wire cutter, diagonal, KS Tools Ergotorque, 115.1012
9	Pliers, Projahn, ISO 5746
10	Pliers, KS Tools, 115.1024
11	Plastic spatula, Kartell
12	Tweezers

The LCA was following the norm ISO14040, including the definition of the goal and scope, the inventory analysis, the impact assessment, and the interpretation. According to the request of Jabra, it was focused on the environmental impact category climate change. Therefore, the indicator "climate change w/o LT, GWP100" from the group "Midpoint (H) w/o LT" of the ReCiPe Method was applied. A hierarchical perspective (H) was selected which is based on scientific consensus regarding time frame and impact mechanisms (Huijbregts et al. 2017). GWP100 expresses the global warming potential over 100 years (Huijbregts et al. 2017) without long-term emissions taken into account (Ecoinvent 2021). The corresponding unit is kg CO_2-Equivalent.

A hotspot analysis was included to identify the life cycle stages and materials with the highest climate impact. The utilized data was based on the results of the product dismantling trials, especially on the bill of materials, and information provided by the manufacturer. Where necessary, additional assumptions were made, explained and justified in 4.2. Result activities (allocation: cut-off) from the database ecoinvent v3.7.1 were used for the evaluation of the environmental impacts of the background processes.

The wireless audio transmission by a headphone during its total lifetime was chosen as the functional unit. The following reference flow resulted from the functional unit:

- One headphone Jabra Evolve2 85 (28599-989-999)
- 2,600 using hours (2a $\times$ 52 weeks/a $\times$ 5 d/week $\times$ 5 h/d)

It is to be noted that the parameter "using hours" will sincerely influence the result which is why the effect of this parameter was to be re-viewed by a sensitivity analysis regarding a repair scenario.

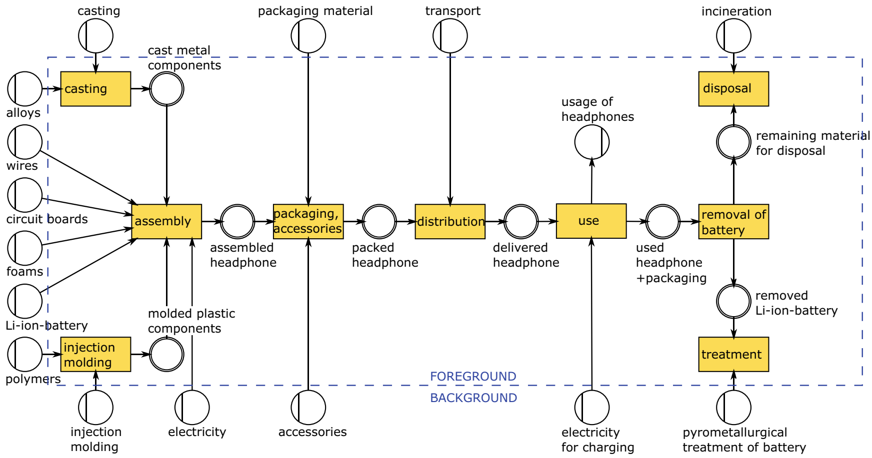


Fig. 2.2 System flow chart

The modeling of the product system and the impact assessment was conducted with the LCA software Umberto LCA + (ifu Hamburg). The system flow chart including foreground and background system identification is displayed in Fig. 2.2.

According to the resources availability in this student research project, the foreground system and its processes were reasonably simplified, but without compromising on the overall correctness of the results.

2.4 Results

2.4.1 Dismantling Trials

In total, 83 dismantling steps were documented. The dismantling hierarchy with all steps including the single required time and the needed tools for every step is displayed in the electronic supplementary material 2.1.

The initial weight of the product was 284.2 g excluding the packaging. Figure 2.3 shows the product before dismantling (left) and after dismantling (right). The mass recovery rate after dismantling was at 98.7% (280.7 g).

The respective masses of the single components and the identification of materials resulted in the bill of materials, displayed in Table 2.2. The product was found to be composed of 61.7% polymers, 20.9% metals, 4.8% circuit boards, 4.6% battery, 3.5% foams, 3.0% wires and 1.7% not identifiable polymers. Regarding the single materials, PC/ABS has the highest mass share with 30.9%, followed by polycarbonate with 16.8%.

Fig. 2.3 Headphone Jabra Evolve2 85 before (left) and after dismantling (right)

2.4.2 Life Cycle Assessment

Inventory analysis. Based on the bill of materials, information given by the manufacturer, and additional assumptions, all relevant input and output flows of the product system were quantified in the inventory analysis. The quantified flows, their corresponding types and the sources of the values are given in Table 2.3.

Simplifications and assumptions to fill data gaps were made as follows: For the processes injection molding and casting, the bill of materials was utilized with a cut-off criterion of <1% of the total product's mass. 50% of the weight of the PC/ABS blend was allocated to polycarbonate and 50% to acylonitrile-butadiene-styrene. It was assumed that all polymers are injection molded and all metals are cast. The electricity needed for assembly was calculated with the total energy consumption in the assembly factory and the model's share of batch size provided by the manufacturer. The masses of an exemplary package, shown in Fig. 2.4, were considered. The weight of the charging stand was not taken into account since it is not delivered with the headphone model 28599-989-999.

The assembly factory is located in Xiamen, China. The distribution in the European Union was to be presented by a combination of the distance from Xiamen to Rotterdam by ship and the average distance from Rotterdam to all EU capitals by lorry. With 2,600 using hours and a time between charges of 21 h with busy light and active noise cancellation on (GN Audio A/S 2020a, b, c), the number of charges was computed. The required electricity per charge is 2.7 Wh as labeled on the battery.

$$\text{Total charging electricity [kWh]} = 2,600\text{ h}/21\text{ h} \times 2,7\text{ Wh} \times 10^{(-3)} = 0.334\text{ kWh} \tag{2.1}$$

Table 2.2 Bill of materials

Material	Weight (g)	%
Circuit boards	13.4	4.8
Wires	8.3	3.0
Lithium-ion-battery	12.9	4.6
Foams	9.7	3.5
Not identifiable polymers	4.7	1.7
Metals	58.6	20.9
Copper alloy	0.2	0.1
Steel alloy	12.6	4.5
Iron-nickel–chromium alloy	20.4	7.3
Nickel alloy	0.2	0.1
Iron alloy	24.8	8.8
Gold alloy	0.4	0.1
Polymers	173.1	61.7
Polyamide	15.3	5.5
Butylformate	0.1	0.0
Polyethylene terephthalate	3.9	1.4
Polymethyl methacrylate	0.2	0.1
Polypropylene	3.9	1.4
Polydimethylsiloxane	0.1	0.0
Polyoxymethylene	0.1	0.0
Polycarbonate	47.1	16.8
PC/ABS	86.6	30.9
Polybutylene terephthalate	15.8	5.6
Sum	280.7	100.0

The end-of-life stage was simplified by assuming that all remaining material after the battery's removal are incinerated. It was assumed that the headphones have a size compatible with residual waste disposal collection bins, although disposal via this route is not compliant with EU rules for WEEE disposal (WEEE Directive 2012). Nonetheless when having reached their end of life, small WEEE may be disposed of via this route, resulting in the current (too low) collection rates e.g. in Germany (Bundesministerium für Umwelt, Naturschutz und nukleare Sicherheit 2019).

Impact assessment. The total global warming potential results in 12.17 kg CO_2-Eq. Table 2.4 shows the contributions allocated to the life cycle stages manufacturing, packaging, distribution, use, and end-of-life.

The manufacturing phase provides 81.2% of the climate impact, which is by far the highest contribution to global warming potential of all headphone life cycle phases, followed by packaging with 10.9%, end-of-life with 3.6%, distribution with 2.8%, and use with 1.5%. The main contribution in the manufacturing phase stems

Table 2.3 Inventory analysis

Material	Unit	Value	Source	Boundary	Type	Input/ output
Injection molding						
Polyamide	g	15.3	Measurement	Extern	Primary product	Input
Polyethylene terephthalate	g	3.9	Measurement	Extern	Primary product	Input
Polypropylene	g	3.9	Measurement	Extern	Primary product	Input
Polybutylene terephthalate	g	15.8	Measurement	Extern	Primary product	Input
Polycarbonate	g	90.4	Calculation	Extern	Primary product	Input
Acrylonitrile–butadiene–styrene	g	43.3	Calculation	Extern	Primary product	Input
Injection molding	g	172.6	Calculation	Extern	Auxiliary and operating materials	Input
Molded plastic components	g	172.6	Calculation	Intern	Intermediate product	Output
Casting						
Steel alloy	g	12.6	Measurement	Extern	primary product	Input
Iron-nickel–chromium alloy	g	20.4	Measurement	Extern	Primary product	Input
Iron alloy	g	24.8	Measurement	Extern	Primary product	Input
Casting	g	57.8	Calculation	Extern	Auxiliary and operating materials	Input
Cast metal components	g	57.8	Calculation	Intern	Intermediate product	Output
Assembly						
Molded plastic components	g	172.6	Measurement	Intern	Intermediate product	Input
Cast metal components	g	57.8	Measurement	Intern	Intermediate product	Input
Li-ion battery	g	12.9	Measurement	Extern	Primary product	Input
Circuit boards	g	13.4	Measurement	Extern	Primary product	Input

(continued)

Table 2.3 (continued)

Material	Unit	Value	Source	Boundary	Type	Input/ output
Wires	g	8.3	Measurement	Extern	Primary product	Input
Foams	g	9.7	Measurement	Extern	Primary product	Input
Electricity	kWh	2.13	Calculation	Extern	Energy	Input
Assembled headphone	Unit	1	Specification	Intern	Intermediate product	Output
Packaging + Adding Accessories						
Assembled headphone	Unit	1	Specification	Intern	Intermediate product	Input
Bag, polyester	g	166.6	Measurement	Extern	Primary product	Input
Carton	g	106.8	Measurement	Extern	Primary product	Input
Paper sleeve	g	55.5	Measurement	Extern	Primary product	Input
Accessories (wires and plugs)	g	41.5	Measurement	Extern	Primary product	Input
Packed headphone	unit	1	Specification	Intern	Intermediate product	Output
Distribution						
Packed headphone	Unit	1	Specification	Intern	Intermediate product	Input
Transport, ship	t*km	14.93	Calculation	Extern	Auxiliary and operating materials	Input
Transport, lorry	t*km	1.25	Calculation	Extern	Auxiliary and operating materials	Input
Delivered headphone	Unit	1	Specification	Intern	Product	Output
Use						
Delivered headphone	Unit	1	Specification	Intern	Product	Input
Electricity for charging	kWh	0.334	Calculation	Extern	Energy	Input
Used headphone + packaging	Unit	1	Specification	Intern	Waste	Output
Usage of headphone	Unit	1	Specification	Reference flow	Reference flow	Output
Removal of battery						
Used headphone + packaging	Unit	1	Specification	Intern	Waste	Input

(continued)

Table 2.3 (continued)

Material	Unit	Value	Source	Boundary	Type	Input/output
Removed Li-ion battery	g	12.9	Measurement	Intern	Waste	Output
Remaining material for disposal	g	804.8	Calculation	Intern	Waste	Output
Disposal						
Remaining material for disposal	g	804.8	Specification	Intern	Waste	Input
Incineration	g	804.8	Calculation	Extern	Auxiliary and operating materials	Input
Treatment						
Removed Li-ion battery	g	12.9	Specification	Intern	Waste	Input
Pyrometallurgical treatment	g	12.9	Calculation	Extern	Auxiliary and operating materials	Input

Fig. 2.4 Exemplary package

from circuit boards, molded plastic components, cast metal components, and the electricity needed for assembly. The polyester bag has the highest share of the GWP in the packaging phase with 66.2%. The transport by ship over the assumed distance of 18,258 km adds 0.14 kg CO_2-Eq and the transport by lorry over the assumed distance of 1,533 km contributes 0.2 kg CO_2-Eq. With the reference flow of 2,600 using hours, the contribution of the electricity for charging is comparatively low with 0.18 kg CO_2-Eq. The incineration of the remaining material after removing the battery has the main share of the GWP during the end-of-life phase with 95.5%.

Sensitivity analysis. The influence of prolonging the product's duration of use on the environmental impact assessment was analyzed by looking at an exemplary repair scenario, the replacement of the battery. It was assumed that the headphone can and will be used for an additional 2,600 h after the replacement of the battery, which results in a total of 0.668 kWh electricity required for charging (0.334 kWh $\times$ 2). The input of the new battery and the treatment of the old battery were considered as well. The scenario was simplified by assuming that the battery can be replaced

Table 2.4 Global warming potential allocated to life cycle stages (kg CO_2-Eq)

Manufacturing	GWP (kg CO_2-Eq)	Packaging	GWP (kg CO_2-Eq)	Distribution	GWP (kg CO_2-Eq)	Use	GWP (kg CO_2-Eq)	End-of-life	GWP (kg CO_2-Eq)
Electricity	1.43	Accessories	0.24	Transport, lorry	0.2	Electricity charging	0.18	Incineration	0.42
Foams	0.05	Bag, polyester	0.88	Transport, ship	0.14			Pyrometallurgic treatment	0.02
Wires	0.05	Carton	0.14						
Circuit boards	4.92	Paper sleeve	0.04						
Li-ion-battery	0.10								
Molded plastic components	1.32								
Cast metal components	2.02								
Total	9.88	Total	1.33	Total	0.34	Total	0.18	Total	0.44

Table 2.5 Repair scenario—total GWP

	GWP (kg CO_2-Eq)
Total GWP—scenario without battery replacement	12.17
+ new Li-ion-battery	0.1
+ additional electricity for charging	0.18
+ pyrometallurgical treatment of first battery	0.02
Total GWP—scenario with battery replacement	12.47

without destruction and without other efforts. The additional contribution to the total GWP for the scenario of the battery replacement is presented in Table 2.5.

To compare the environmental impact of the initial scenario and the repair scenario, the corresponding values for the total global warming potential were placed in relation to the two different assumptions for the using hours. Prolonging the assumed duration of use of the product by the exemplary repair scenario decreases the assessed global warming potential per hour from 4.7 g CO_2-Eq/h to 2.4 g CO_2-Eq/h. This effect is clearly caused by the extremely high climate relevance of the manufacturing phase of the product.

2.5 Discussion

The product Jabra Evolve2 85 was found to be composed of 61.7% polymers, 20.9% metals, 4.8% circuit boards, 4.6% battery, 3.5% foams, 3.0% wires and 1.7% not identifiable polymers. Many polymer components were not identifiable via FTIR-ATR due to their dark color, so the manufacturer's material marks had to be used for material identification. If material recycling in the end-of-life phase is intended in future, appropriate light colours could improve identification for mechanical recycling.

The GWP contribution of the Jabra Evolve2 85 of 12.17 kg CO_2-Eq can be compared to the result of an impact assessment of the materials of the Jabra Biz 1500 Mono QD, conducted by Master students at Pforzheim university. The product is a lightweight design, mono speaker headset for business use and can be taken as an extreme example of material saving. The assessed global warming potential of the 61 g mono-headset was 2.51 kg CO_2-Eq (Melter and Mai 2021), which equals around 21% of the GWP of the stereo headset Jabra Evolve2 85. The GWP per gram product is comparable with 41.1 g CO_2-Eq/g product for the lightweight mono speaker headset and 42.8 g CO_2-Eq/g for the stereo headset.

The result of the LCA entails some uncertainties due to simplifications and assumptions: In the inventory analysis, the manufacturing processes of the single materials could be considered in more detail, as in the current study all polymers were assumed to be injection molded and all alloys to be cast. Depending on the selected means of transport, the expected sales market and the markets distance to the incoming shipping port, the contribution of the distribution phase can vary. Other scenarios for the end-of-life phase could be developed, mainly the consideration of recycling as required by the WEEE regulation (WEEE Directive 2012; ElektroG 2015). As current collection rates are very low, the actual disposal way for not separately collected WEEE is unknown (Kummer et al. 2021), and headphones may geometrically fit into the residual waste collection bins, the disposal via residual waste was assumed as realistic for this study, although not legal.

The result of the impact assessment depends on the selected background data sets, e.g. the electricity mix for manufacturing. In some cases, no exactly matching data sets in ecoinvent were available which required adjustments and assumptions, for example the ecoinvent activity “market for textile, non-woven polyester” for the bag fabrics. For further improvement of data quality, additional data bases or other better-matching data sets could be utilized. The value of the lifetime of two years was based on manufacturer’s specification. Additional market studies or surveys to provide data for further scenarios with different values could be conducted in future. The repair scenario used was highly simplified by assuming that the using hours of 2,600 h are doubling by repair. However, the basic finding that prolonging the lifetime by repair can be seen as environmentally beneficial can be regarded as valid for products with comparable low environmental impact in the use phase (Bovea et al. 2020; Hischier and Böni 2021; Pamminger et al. 2021).

2.6 Conclusion

By dismantling, ear cases, earpads, speakers, main circuit boards, the microphone arm, the battery, and the headband were assumed as main components. 98.3% of all materials’ weight could be identified via manufacturer’s marks, ATR and XRF. The identified materials could be categorized in polymers (61.7%), metals (20.9%), circuit boards (4.8%), Li-ion-battery (4.6%), foams (3.5%) and wires (3.0%). 97.9% of all materials’ weight were used as input for the life cycle assessment. With a reference flow of one headphone and 2,600 using hours, the impact assessment of a Jabra Evolve2 85 headphone resulted in a total climate impact of 12.17 kg CO_2-Eq. This result of global warming potential covered all the relevant life cycle phases manufacturing, packaging, distribution, use, and end-of-life. Manufacturing shows by far the highest climate contribution with a total of 9.88 kg CO_2-Eq and a contribution of materials of 6.40 kg CO_2-Eq. Considering the repair scenario with a battery replacement, repairability and prolonging the lifetime can have a significant impact on the global warming potential per hour.

Acknowledgements The authors thank Maurits Hekking, Albert Brønd, and Catharina Cold from GN Store Nord A/S and Michael Hoffmann from TechProtect GmbH for the fruitful collaboration and for providing exemplary products and information. Special thanks to Alvaro Cota Albo for the thriving cooperation. The exchange of information with Yannik Melter, Christian Mai, Sara Fileppi, Henrik Neumann, Markus Baisch, Master students at Pforzheim University, is gratefully acknowledged.

References

Audio-technica (n.d.) ATH-M50xBT. https://www.audiotechnica.com/de-de/ath-m50xbt. Accessed 14 Mar 2021

Bang & Olufsen (n.d.) B&O Beoplay H9 3rd Gen. https://www.bang-olufsen.com/de/kopfhoerer/beoplay-h9?variant=beoplay-h9-3-matte-black. Accessed 14 Mar 2021

Bovea MD et al (2020) Repair vs. replacement: Selection of the best end-of-life scenario for small household electric and electronic equipment based on life cycle assessment. J Environ Manage 254:109679

Broussely M et al (2001) Aging mechanism in Li ion cells and calendar life predictions. J Power Sources 97–98(4):13–21

Bundesministerium für Umwelt, Naturschutz und nukleare Sicherheit (2019) Berichtspflicht gemäß Art. 16 Absatz 4 der Richtlinie 2012/19/EU über Elektro- und Elektronikaltgeräte (WEEE-Richtlinie) – Berichtsjahr 2019. https://www.bmuv.de/fileadmin/Daten_BMU/Download_PDF/Abfallwirtschaft/elektronikgeraete_daten_2019_bf.pdf. Accessed 20 Dez

Ecoinvent (2021) What does LT and w/o LT mean? https://www.ecoinvent.org/support/faqs/methodology-of-ecoinvent-3/what-does-lt-and-wo-lt-mean.html. Accessed 12 May 2021

ElektroG (2015) Gesetz über das Inverkehrbringen, die Rücknahme und die umweltverträgliche Entsorgung von Elektro- und Elektronikgeräten

Gfu Consumer & Home Electronics GmbH and Growth from Knowledge (2020), Home Electronics Markt Index Deutschland (HEMIX): Januar 2020 - Dezember 2020. https://gfu.de/markt-zahlen/hemix-2020/. Accessed 08 Nov 2021

GN Audio A/S (n.d.a) Jabras Grüne initiative—recycle und profitiere. https://green.jabra.com/start. Accessed 07 Nov 2021

GN Audio A/S (n.d.b) Ohrpolster. https://www.jabra.com.de/accessories#?pfs=discountedprice-desc&pff=bc59. Accessed 09 May 2021

GN Audio A/S (2019) Jabra Elite 85h. https://www.jabra.com.de/bluetooth-headsets/jabra-elite-85h. Accessed 14 Mar 2021

GN Audio A/S (2020a) GN sustainability ESG report. https://www.gn.com/about/corporate-sustainability. Accessed 07 Nov 2021

GN Audio A/S (2020b) Jabra Evolve 2 85. https://www.jabra.com.de/business/office-headsets/jabra-evolve/jabra-evolve2-85. Accessed 23 Feb 2022

GN Audio A/S (2020c) Jabra evolve 65. https://www.jabra.com.de/business/office-headsets/jabra-evolve/jabra-evolve-65. Accessed 14 Mar 2021

Harman International Industries Inc. (2019) JBL Live650BTN. https://de.jbl.com/on/demandware.static/-/Sites-masterCatalog_Harman/default/dw57501e8a/pdfs/JBL_Live650BTNC_Spec_Sheet_German.pdf. Accessed 14 Mar 2021

Hischier R, Böni HW (2021) Combining environmental and economic factors to evaluate the reuse of electrical and electronic equipment—a Swiss case study. Resour Conserv Recycl 166:105307

Huijbregts M et al (2017) ReCiPe 2016 v1.1: a harmonized life cycle impact assessment method at midpoint and endpoint level, Report I: Characterization. Bilthoven, Netherlands

Job R (2020) Electrochemical energy storage: physics and chemistry of batteries. De Gruyter, Beaverton (United States)

Korthauer R (2013) Handbuch Lithium-Ionen-Batterien. Springer, Berlin Heidelberg (Germany)
Kummer S et al (2021) Consumer survey on the final consumer behavior concerning the disposal of WEEE in Germany. Waste Manage Res J Int Solid Wastes Public Cleansing Assoc 39:1–7
Lautsprecher Teufel GmbH (n.d.) Teufel Supreme on. https://teufel.de/supreme-on-105957000#pdp-specifications. Accessed 14 Mar 2021
Logitech (n.d.) Logitech zone wireless. https://www.logitech.com/de-de/products/headsets/zone-wireless-bluetooth.html?crid=1875. Accessed 18 May 2021
Maia LK et al (2019) Expanding the lifetime of Li-ion batteries through optimization of charging profiles. J Clean Prod 225:928–938
Melter Y, Mai C (2021) Circular economy approaches for headphones based on the example of mono-headsets Jabra biz 1500 mono QD. Poster
Pamminger R et al (2021) Modelling of different circular end-of-use scenarios for smartphones. Int J Life Cycle Assess 26:470–482
Panasonic Deutschland (n.d.) RP-HD610N. https://www.panasonic.com/de/consumer/home-entertainment/kopf-ohrhoerer/kopfhoerer/rp-hd610n.html. Accessed 17 May 2021
Plantronics Inc. (2021) Voyager 4200 Office und UC. https://www.poly.com/content/dam/plantronics/documents-and-guides/product-sheets/voyager-4200-uc-ps-de.pdf Accessed 10 Jul 2022
Schulze R, Buchert M (2016) Estimates of global REE recycling potentials from NdFeB magnet material. Resour Conserv Recycl 113:12–27
Sony Europe B.V. (n.d.) Sony WH-1000xXM3, https://www.sony.de/electronics/support/wireless-headphones-bluetooth-headphones/wh-1000xm3/specifications. Accessed 14 May 2021
Thiébaud-Müller E et al. (2018) Service lifetime, storage time, and disposal pathways of electronic equipment: a Swiss case study: supporting information 1. Journal of Industrial Ecology 22(1): 196–208
WEEE Directive (2012) Directive 2012/19/EU of the European parliament and of the council of 4 July 2012 on waste electrical and electronic equipment

Chapter 3
Implementing Ecodesign During Product Development: An Ex-Ante Life Cycle Assessment of Wood-Plastic Composites

Nadine Brunnhuber, Andreas Windsperger, Enrique Alejandro Perdomo Echenique, and Franziska Hesser

Abstract About 80% of environmental performance is determined during product development. This study assesses environmental impacts of wood-plastic composite (WPC) boards still in development, to identify impact hot spots and improvement potentials. A seven-step approach to ecodesign implementation was used. It identifies environmental impacts and derives improvement strategies. A life cycle assessment (LCA) according to ISO 14040 was conducted to quantify potential environmental impacts. The WPC boards are made of PVC and wood flour. Impacts mostly result from PVC and electricity consumption for production. Thus, this study proposes replacing PVC with polylactic acid (PLA). Further improvement strategies are increasing material efficiency, energy efficiency, renewable electricity use and secondary plastic input. Increased end of life recycling reduces environmental impacts, compared to incineration only. These changes reduce the initial climate change results of 145 kg CO_2 eq by 55%. Thus, early consideration of environmental aspects supports sustainable product development.

Keywords Life cycle assessment · Ecodesign · Ex-ante · Wood-plastic composite · PLA

N. Brunnhuber (✉) · A. Windsperger
Institute for Industrial Ecology, Rennbahnstraße 29, 3100 St. Pölten, Austria
e-mail: nadine.brunnhuber@indoek.at

A. Windsperger
Department of Material Sciences and Process Engineering, Institute of Wood Technology and Renewable Materials, University of Natural Resources and Life Sciences, Konrad Lorenz-Straße 24, Tulln an Der Donau, 3430 Vienna, Austria

E. A. Perdomo Echenique
Department of Economics and Social Sciences, Institute of Marketing and Innovation, University of Natural Resources and Life Sciences, Feistmantelstraße 4, 1180 Vienna, Austria

E. A. Perdomo Echenique · F. Hesser
Wood K plus—Competence Centre for Wood Composites and Wood Chemistry, Kompetenzzentrum Holz GmbH, Altenberger Straße 69, 4040 Linz, Austria

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*,
Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_3

3.1 Introduction

About 80% of a product's environmental performance is determined during product development (McAloone and Bey 2009). Thus, considering environmental impacts throughout a product's life cycle beginning during its development is necessary to create a more sustainable product. The ecodesign concept provides a framework for this, as it aims at minimizing environmental impacts without compromising other essential factors (e.g., cost, quality, performance) by integrating environmental considerations into the product development process (van Weenen 1995). Incorporating ecodesign into business processes results in improved product development processes (Rodrigues et al. 2018). This enhances both sustainability performance (Rodrigues et al. 2018) and business outcomes (Pigosso et al. 2013). Traditional ecodesign approaches in product development focus on either environmental evaluations (LCA) or using guidelines to environmentally improve product design (Tchertchian et al. 2013). This study combines both of these approaches by assessing the environmental impacts starting during product development and then optimizing the product's environmental performance in accordance with the ecodesign framework. This study addresses the question of how the material composition and the use of secondary raw materials affect the WPC boards' environmental performance. Additionally, the potential of ex-ante LCA to implement ecodesign during the development stage of WPC boards is assessed.

Studies on combining LCA and ecodesign are not new. González-García et al. (2012) conducted an LCA on a wooden modular playground and created ecodesign strategies to reduce environmental impacts. Gutiérrez Aguilar et al. (2017) reduced environmental impacts of a wooden chair by incorporating LCA into product design. Cobut et al. (2015) created ecodesign strategies based on an LCA and scenario analysis of wooden doors. However, no case studies on the seven-step approach have been published so far. Through the seven-step approach, this study combines ex-ante LCA and ecodesign thinking to steer towards sustainable product design starting during the material development stage. Despite its negative effects on the environment and human health (Bidoki and Wittlinger 2010), global PVC use continues to grow (Markarian 2007). PVC has a high chlorine content (Sarker et al. 2012). As this causes a number environmental issues (Thornton s.a.), PVC is considered unsustainable (Leadbitter 2002). However, because of the presence of chlorine, PVC is very durable in use (Leadbitter 2002). Due to its unique properties, PVC is widely used in the construction sector (e.g., pipes, window frames) (Markarian 2007), In construction, it replaces traditional building materials like wood (Bidoki and Wittlinger 2010). In some cases (e.g., wall claddings, floor decking), wood-plastic composites (WPC) can be used instead of neat plastic or solid wood products (Sommerhuber et al. 2017).

According to EN 15,534-1 (European Committee for Standardization 2014), WPCs are materials made of a combination of one or more cellulose-based materials and one or more thermoplastics, which are then processed through plastic processing techniques (European Committee for Standardization 2014). This study uses LCA to assess the environmental performance of WPC flooring boards made of PVC and wood flour, which are still in product development. The wood flour consists of beech wood saw dust and poplar bark chips.

3.2 Methods

The seven-step approach by McAloone and Bey (2009) provides guidance on integrating ecodesign thinking into product development and incorporates an LCA in accordance with ISO 14040. The seven consecutive steps aim at giving an overview of environmental impacts of the product or service investigated, creating concepts for environmental improvements and deriving proposals for an environmental product development strategy. During this study, an ex-ante LCA is conducted. This type of LCA is defined as "performing an environmental LCA of a new technology before it is commercially implemented in order to guide R&D decisions to make this new technology environmentally competitive with the incumbent technology mix" (van der Giesen et al. 2020).

3.2.1 The Seven Steps Towards Ecodesign

There are many different methodologies for integrating environmental aspects into product design (Bovea and Pérez-Belis 2012). The seven-step approach was chosen because of its transdisciplinary nature, as it combines qualitative and quantitative methods. Furthermore, it can be applied early on during product development. Figure 3.1 shows a graphical overview of the seven consecutive steps.

In step 1, the product's use context and functionality to the user are established. This provides the benchmark for following decisions or comparing alternative concepts. Environmental impacts associated with the product are identified for each

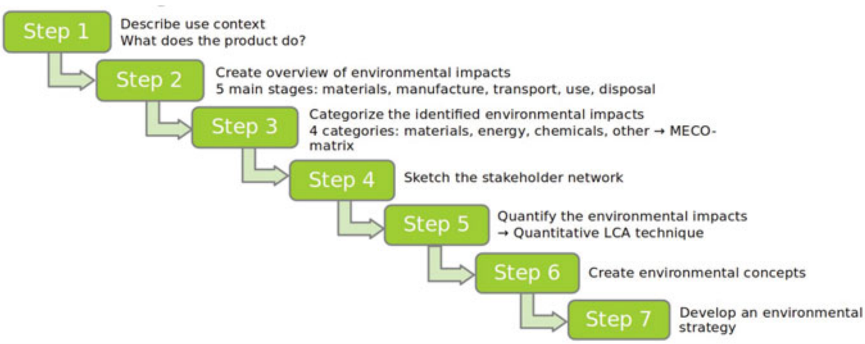


Fig. 3.1 Graphical representation of the seven-step approach by McAloone and Bey (2009) (own illustration)

life cycle stage in step 2. This is done through a review of scientific literature. In step 3, these impacts are assigned to one of four categories: materials, energy, chemicals or other. This results in the so-called MECO-matrix. It provides an overview of where environmental impacts occur along the life cycle and their root causes. This helps identify and prioritize environmental focus areas for improvement. Steps 1 to 3 give a qualitative overview of the product life cycle and its impact hot spots. This helps identify the environmental focus areas and serves as a preparation for the goal and scope definition of the LCA. Additionally, aspects which do not contribute to the product's environmental impacts (e.g., use phase of the WPC boards) are identified and can then be cut-off in the LCA.

During step 4, a sketch of the stakeholder network is created. This includes all stakeholders relevant to the product's life cycle, as well as material flows and information exchanges between them (McAloone and Bey 2009). This is important because environmental impacts often occur in stakeholder exchanges (e.g., negotiations along the supply chain) (McAloone and Pigosso 2018). The environmental impacts considered the most substantial are highlighted in the stakeholder network sketch. This way, key stakeholders who need to be included in considerations for environmental improvement are identified.

Environmental impacts are quantified by a LCA in step 5. This way, the extent of environmental impacts is determined, and later impact reductions are measurable. Based on the results from the previous steps and the LCA results, environmental concepts are created in step 6. This means that new product concepts, which provide the same functionality by innovative and improved ways, are created based on selected ecodesign principles. These selected ecodesign principles are as follows:

- Reduction of the material intensity of the product
- Reduction of the energy intensity of the product
- Reduction of the dispersion of harmful substances through the product
- Increase in amount of recycled and recyclable materials in the product
- Maximization of the use of sustainable resources and supply chains

This step develops approaches to eliminate the problem causing the environmental impacts. In step 7, environmental strategies are derived from the results of the previous steps. These strategies are quantitative goals for environmental improvements to become rooted in the entire organization. However, this study only created strategies on a product-level.

3.2.2 LCA Methodology

An LCA in accordance with ISO 14040 is conducted to quantify the environmental impacts. Because the studied product is still in product development, this LCA can be classified as ex-ante (Buyle et al. 2019). The LCA methodology is elaborated in detail in the following subchapters.

Goal and scope definition. The goal of this study is to identify environmental impacts, impact hot spots, and impact reduction potentials for the WPC flooring boards. The functional unit is the provision of walkability of 1 m^2 outdoor terrace flooring over a time span of 30 years (Verband der Deutschen Holzwerkstoffindustrie e.V. (VHI) 2015). The reference flow providing the functional unit is 29.86 kg WPC flooring boards. The system boundary is cradle to grave. The LCA was conducted with the open source software openLCA (version 1.10.3).

Inventory analysis. The WPC boards consist of PVC (44.45 wt%), wood flour (44.45 wt%), CaZn-based stabilizer (5.33 wt%), calcium carbonate (4.44 wt%), and masterbatch (1.33 wt%). Because of a lack of data availability, the stabilizer and masterbatch were not considered in this study. The product system is illustrated in Fig. 3.2. The wood flour is manufactured in Slovakia. It consists of 75 wt% beech wood flour from a saw mill and 25 wt% poplar bark chips. The poplar bark originates from the EU-funded Dendromass4Europe short rotation coppice (SRC) project, which aims at establishing regional value chains for new bio-based materials (Dendromass4Europe 2021). The wood flour is transported to the Czech Republic, where the WPC boards are produced. Through extrusion, wood flour, PVC, and additives are manufactured into WPC boards. Then, the WPC boards are packed onto wooden pallets. The pallet input was created with input data from Deviatkin et al. (2019). The WPC boards are sold to the customer, who uses them for terrace flooring. Due to the small amounts of input per functional unit, installation and maintenance (e.g., water and cleaning agents) were not considered in this assessment. At the end of the WPC boards' use phase, the WPC board waste is collected. Incineration was assumed as end of life (EoL) treatment for WPC boards and wooden pallets. For waste incineration, a transport distance for waste collection of 30 km was assumed (Gantner 2012).

D4EU contributed material and energy input data for poplar bark production. The WPC manufacturer (Energochemica SE) provided input data for wood flour and WPC production. Table 3.1 lists the life cycle inventory data for the WPC production system. Additional data from scientific sources were used. Background data for supply processes were taken from the European Reference Life Cycle Database (ELCD database version 3.2). This database was used because it is publically available at the openLCA nexus website and provides extensive life cycle inventory data, specifically for Europe.

Impact assessment. The impact assessment method used is ILCD 2011 midpoint+. The impact categories selected for this study are acidification (AP), climate change (GWP), freshwater ecotoxicity (FRWTOX), freshwater eutrophication (EUTF), human toxicity, cancer effects (HTCE), land use (LU), particulate matter (PM), and water resource depletion (WD). This selection is based on the impact categories deemed relevant to the wood furniture sector (Bianco et al. 2021). This assessment does not include any normalization or weighting.

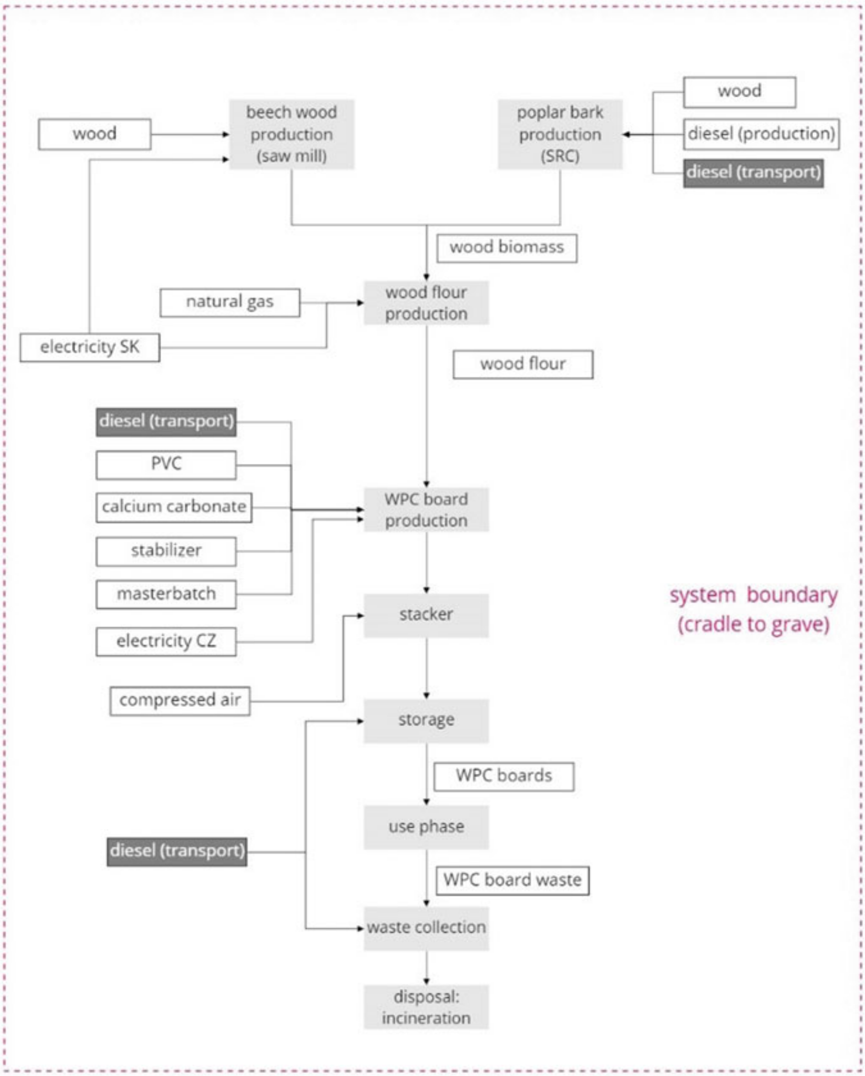


Fig. 3.2 Simplified representation of the WPC board product system within the system boundary

3.3 Results

The results are presented following the seven steps of the approach to provide a clear overview to the reader. In addition to the seven-step approach, the results of implementing the environmental strategies into the product life cycle are presented. The quantitative improvements through this ecodesign product optimization are given at the end of this chapter.

Table 3.1 Life cycle inventory inputs per reference flow for the WPC board life cycle

Input	Quantity	Unit	Source
Calcium carbonate	1.33	kg	Energochemica (2021)
Compressed air	1.49E−05	m^3	Energochemica (2021)
Diesel (transport of final WPC boards to retailer and customer)	0.26	kg	Perdomo et al. (2021), Energochemica (2021)
Electricity (CZ grid mix)	40.58	kWh	Energochemica (2021)
Electricity (SK grid mix)	1.03	kWh	Energochemica (2021)
Hard wood (beech wood)	9.96	kg	Perdomo et al. (2021), Energochemica (2021)
Hard wood (poplar biomass needed to achieve the poplar bark content)	19.52	kg	Clark and Schroeder (1977)
Occupation, forest, primary (non-use)	0.92	m^2	Perdomo et al. (2021)
Natural gas	0.60	kg	Energochemica (2021)
PVC	13.27	kg	Energochemica (2021)
Cooling water	2.24	m^3	Energochemica (2021)
Wooden pallet	0.73	kg	Deviatkin et al. (2019)
Lorry transport of wood flour to WPC production facility	13.242*440	kg*km	Energochemica (2021)
Lorry transport of WPC board waste	29.86*30	kg*km	Gantner (2012)
Output	Quantity	Unit	Source
WPC boards	29.86	kg	Energochemica (2021)
Cooling water	2.24	m^3	Energochemica (2021)
Energy from waste incineration	17.38	kWh	ELCD 3.2 (2016)

3.3.1 Step 1: Use Context

The WPC boards are used for outdoor terrace flooring over a life span of 30 years. Households and household-like organizations in the eastern European Union are their users. After their use, the WPC boards are discarded and replaced.

3.3.2 Step 2 and 3: Environmental Impacts

The WPC boards' environmental impacts organized by life cycle stage and category are shown in the MECO-matrix in Table 3.2. In the material stage, wood biomass extraction causes loss of wildlife habitat and biodiversity (Higgins 2011). PVC is the emission source of various toxic chemicals, e.g., dioxins or phthalate plasticizers. Many of them have neurotoxic and cancerogenic effects (Thornton s.a.). They often do not biodegrade and accumulate in the biosphere (Allsopp et al.

Table 3.2 MECO-matrix categorizing the environmental impacts associated with the WPC boards' life cycle

	Materials	Manufacturing	Transport	Use phase	Disposal
Materials	• Loss of wildlife habitat and biodiversity				
Energy		• Loss of wildlife habitat and biodiversity • Land use change			
Chemicals	• Ecotoxic effects • Bioaccumulation of pollutants • Health issues (e.g., due to carcinogens, neurotoxins)	• Global warming • Acid rain • Smog • Ecotoxic effects • Contamination of water and soil	• Global warming • Health issues • Acid rain • Smog		• Global warming • Contamination of water and soil • Health issues
Other	• Health risks for workers	• Health risks for workers	• Health risks for workers		• Health risks for workers

2001). Manufacturing-related impacts result from electricity consumption. Electricity generation from fossil fuels causes toxic emissions, which can result in cancer or other health issues (IAEA 1999). CO_2 emissions from fossil fuel combustion contribute to global warming, while SO2 and NOX emissions cause acid rain formation (Alberta Environment 2001). Land use changes through mining activities, and power plant generation causes loss of wildlife habitat and biodiversity (IAEA 1999). Transport contributes to global warming and health issues (e.g., through air pollution, traffic accidents) (Browne et al. 2012). Air pollutants from fossil fuel combustion cause smog formation (Elsom 1996) and acid rain (Alberta Environment 2001). As for this study, no use related environmental impacts were identified. Disposal related impacts are caused by waste incineration. CO_2 emissions contribute to global warming (Pivato et al. 2018). Human health issues (e.g., cancer, respiratory diseases) result from emitted chemicals (Hamer 2003). Incineration residues are landfilled and can contaminate soil or groundwater (Allsopp et al. 2001).

3.3.3 Step 4: Stakeholder Network

Figure 3.3 shows the stakeholder network for the WPC board life cycle. Material flows occur between suppliers and manufacturers via a transportation company. The WPC boards are then sold by a retailer to the customer. WPC board waste is incinerated. Energy is recovered and fed into the electricity grid, which supplies manufacturing processes again. Information flows occur between D4EU and the poplar bark supplier

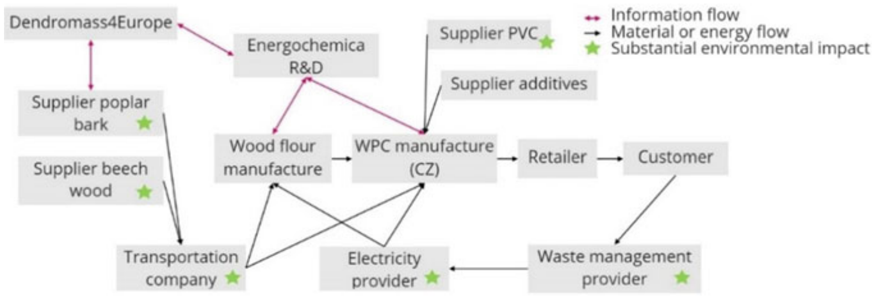


Fig. 3.3 Stakeholder network for the WPC board life cycle including material, energy, and information flows

(D4EU SRC project), as well as between Energochemica R&D and the wood flour and WPC manufacturers. As visible in Fig. 3.3, environmental impacts occur in relation to materials used (PVC and wood biomass), transport, energy generation, and disposal.

3.3.4 Step 5: LCA Results

Acidification associated with the WPC board life cycle is 0.508 mol H + eq. Climate change contribution equals 144.990 kg CO_2 eq. Freshwater ecotoxicity is 3.851 CTUe and freshwater eutrophication is 0.001 kg P eq. The result for human toxicity (cancer effects) is 1.828E-07 CTUh. Land use equals 7.696 kg C deficit. Particulate matter formation associated with the WPC board is 0.029 kg PM2.5 eq. Water resource depletion is 0.565 m^3 water eq.

Figure 3.4 illustrates the contribution of the different life cycle stages to each impact category. The majority of freshwater eutrophication (97%) and human toxicity cancer effects (82%) is material related. With 44%, material is also the main cause of freshwater ecotoxicity. Material impacts result from the PVC. Land use is an exception for this, as 24% of land use impacts result from biomass production. Manufacture is responsible for 54% of acidification, 76% of land use, 59% of particulate matter and 71% of water resource depletion. Electricity consumption is the cause of manufacture impacts. Disposal is responsible for 51% of climate change results. This is caused by end of life incineration of the WPC boards. Transport only contributes very little to the total environmental impacts. No impacts result from the use phase. Thus, the impact hot spots are material, manufacture and disposal. This coincides with the results from steps 3 and 4 (see Table 3.2 and Fig. 3.3). Levers for impact reduction are the use of PVC, electricity consumption and EoL incineration.

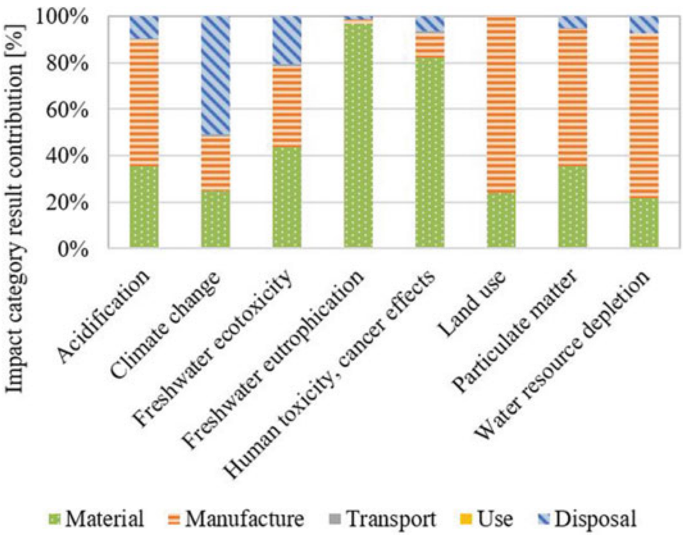


Fig. 3.4 Life cycle stage contribution [%] to impact assessment result by impact category

3.3.5 Step 6: Environmental Concepts

The environmental concepts are based on the ecodesign principles listed by the seven step approach (McAloone and Bey 2009). However, it is impossible to fulfil all of them in one product (McAloone and Pigosso 2018). Figure 3.5 illustrates how LCA findings connect to the ecodesign principles selected for implementation. These environmental concepts are theoretical concepts.

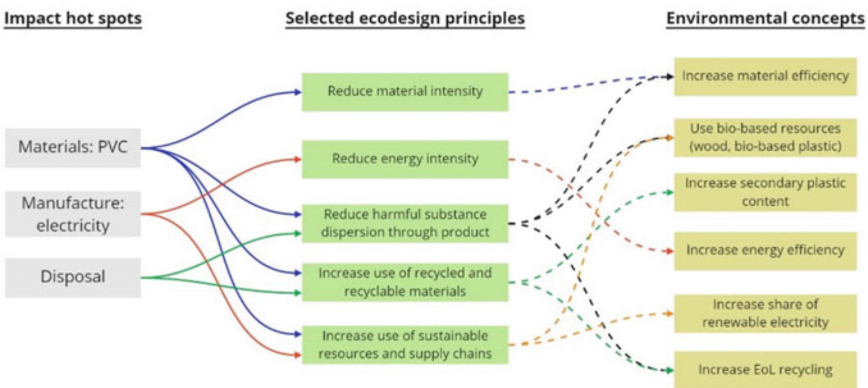


Fig. 3.5 Connections between impact hot spots from LCA results, ecodesign principles, and the derived environmental concepts

Material related impacts are addressed by the principle of reducing material intensity. With increased material efficiency, material related impacts decrease. Reducing the dispersion of harmful substances through the WPC boards also is beneficial for material impacts. This can be achieved by increasing material efficiency and using bio-based resources (e.g., wood biomass, bio-based plastic like PLA). The principle of using recycled and recyclable materials is put into practice by increasing the secondary plastic content and increasing EoL recycling to provide secondary WPC material, which can be used for WPC board manufacturing.

Manufacturing impacts are addressed by improving energy efficiency and reducing energy intensity of WPC board production. The principle of increased use of sustainable resources is implemented by increasing the share of renewable electricity used for WPC board production. Disposal aspects relate to the dispersion of harmful substances through the WPC boards. Increased EoL recycling avoids waste incineration and thus also the generation of toxic incineration residues. This reduces environmental impacts of WPC board disposal. Increased EoL recycling reduces the amount of WPC boards incinerated. Therefore, disposal related environmental impacts are reduced.

3.3.6 Step 7: Environmental Strategies

Figure 3.6 shows the quantitative environmental strategies derived from the environmental concepts. A 20% increase in material efficiency of WPC board production is proposed based on the Roadmap to a Resource Efficient Europe (European Commission 2011). This means that through improving the board design, less WPC material is needed to produce the minimum requisite number of WPC boards to provide 1 m^2 of WPC board flooring. Replacing PVC with a bio-based plastic reduces material related impacts. The bio-based plastic alternative chosen in this study is PLA because it is suitable for WPC production (Kim and Pal 2011). Substituting PVC with PLA results in a PLA content of 44.45 wt%. Using 30 wt% secondary PLA is deemed feasible (Petchwattana et al. 2012).

Based on the EU Clean Energy Package (European Commission 2010), an energy efficiency increase of 20% is proposed. The current renewable electricity share in the Czech grid mix is 14% (Eurostat 2021). This study proposes an increase to 32% for WPC board production. This is based on the EU's renewable energy goals in the Renewable Energy Directive (2009/28/EC) (European Commission 2021). There are no numbers on current WPC recycling rates and no estimations on future WPC recycling rates are available. Recycling 65% of WPC boards is proposed based on municipal waste recycling goals of the European Waste Framework Directive (2008/98/EC) (European Commission 2015). Currently, 38% of PVC post-consumer waste is recycled (PlasticsEurope 2019). The PVC recycling rate is considered because WPC waste can be considered as post-consumer plastic waste (Sommerhuber et al. 2015).

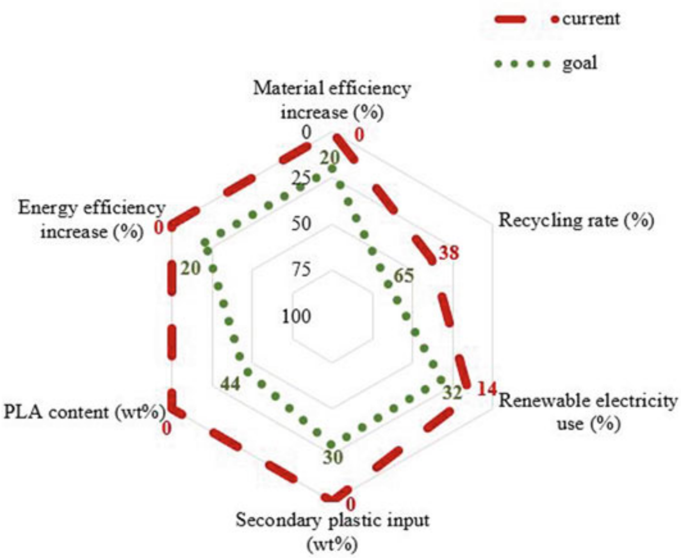


Fig. 3.6 Quantitative environmental strategies and status quo of WPC board production

3.3.7 Ecodesign Product Optimization

Impact category result changes through step-by-step implementation of environmental strategies are shown in Fig. 3.7. Total impact assessment results for the WPC boards after ecodesign implementation are shown in Table 3.3. First, impacts for substitution of PVC with PLA were calculated. Life cycle inventory data for PLA production were taken from scientific literature (Groot and Borén 2010; Vink and Davies 2015). Then, material efficiency, secondary plastic input, energy efficiency and renewable electricity use associated with the life cycle of 1 m^2 of WPC boards were increased. Life cycle inventory data from Stichnothe and Azapagic (2013) were used to create the secondary plastic supplier process. This study assumes that 30% of the primary plastic was replaced with secondary plastic (Cordella et al. 2020; Petchwattana et al. 2012).

Finally, results for a recycling rate of 62% were calculated. Wood particles and thermoplastic matrix of the WPC are irreversibly bonded and thus cannot be separated at the EoL (Sommerhuber et al. 2017). Because of the thermoplastic, WPC materials can be re-extruded to new WPC products (Boeglin et al. 1997). This study refers to this as recycling WPC. The wood content allows for the WPC boards to be recycled into wood particle boards (Schirp 2021). For this study, it is assumed that equal parts of WPC board waste are recycled to WPC and to wood particle boards (31% of total WPC boards each). For recycling to WPC, inventory data from Stichnothe and Azapagic (2013) were used. Recycling to particle boards was modeled using

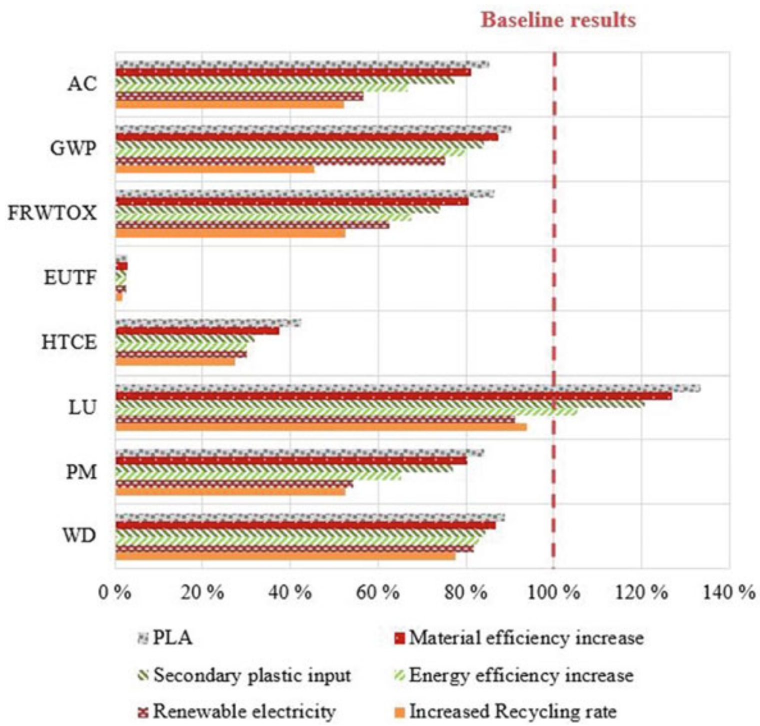


Fig. 3.7 Changes in environmental impacts of the WPC board life cycle after step-by-step implementation of the proposed environmental strategies

Table 3.3 Environmental impacts for the initial WPC boards and the WPC boards after ecodesign implementation

Impact category	Reference Unit	Initial WPC boards	Ecodesign WPC boards	Impact result change [%]
AC	mol H + eq	0.508	0.265	−47.86
GWP	kg CO_2 eq	144.990	65.931	−54.53
FRWTOX	CTUe	3.851	2.023	−47.47
EUTF	kg P eq	0.001	0.000	−98.17
HTCE	CTUh	1.828E−07	4.996E−08	−72.67
LU	kg C deficit	7.696	7.323	−6.03
PM	kg PM2.5 eq	0.029	0.015	−47.42
WD	m^3 water eq	0.565	0.439	−22.29

AC = acidification, GWP = global warming potential, FRWTOX = freshwater ecotoxicity, EUTF = freshwater eutrophication, HTCE = human toxicity (cancer effects), LU = land use, PM = particulate matter, WD = water resource depletion inventory data from Rivela et al. (2006). A distance of 250 km was assumed for waste collection and transport to the recycling facility (Gantner 2012).

As a result of ecodesign implementation, acidification is reduced by 48%. The highest impact reductions were achieved by replacing PVC with PLA and increasing the share of renewable electricity. PLA is responsible for an impact reduction of 15%. Climate change contribution is reduced by 55%, in which increased EoL recycling plays a key role. PLA reduces climate change impacts by 10%. Freshwater ecotoxicity decreases by 47%. Main contributors to this impact reduction are PLA and increased EoL recycling. Freshwater eutrophication is reduced by 98%, mostly because of replacing PVC with PLA (97%). Human toxicity (cancer effects) drops by 73%. Again, PLA is responsible for most of this impact reduction (57%). Total land use results in a 6% decrease. Replacing PVC with PLA increases the land use associated with the WPC boards. Major impact decreases are achieved by increasing energy efficiency and a higher share of renewable electricity. Particulate matter formation decreases by 47%, mostly because of switching to PLA, increasing energy efficiency and increasing renewable electricity consumption. Water resource depletion is reduced by 22%. Again, replacing PVC with PLA strongly contributes to this.

3.4 Discussion

This study identified impact reduction potentials derived improvement strategies in accordance with ecodesign principles. The impact hot spots in the WPC board life cycle are material (PVC), manufacturing (electricity consumption), and disposal (EoL incineration). They offer the biggest potential for environmental improvement. Replacing PVC with a different plastic material, like PLA, reduces environmental impacts. Partly substituting primary plastic with secondary plastic also reduces material impacts. Both recycling to WPC and recycling to wood particle boards decrease disposal related impacts from WPC waste incineration. Thus, the environmental strategies created to improve the WPC boards' environmental performance incorporate increasing material efficiency, energy efficiency, secondary plastic input, share of renewable electricity, and EoL recycling, as well as replacing PVC with PLA.

This is not new; however, most studies on environmentally improving products with the help of LCA and the ecodesign approach only focus on improving product design, e.g., material intensity (Gutiérrez Aguilar et al. 2017) or selection of raw materials (González-García et al. 2012). This study's contribution to sustainability research is improving the WPC boards' material composition and their entire life cycle, based on LCA results and the derived environmental strategies. As a result, not only is the product improved (e.g., reduced impacts through optimized material composition) but also the production process results in less environmental impacts (e.g., through increased energy efficiency and renewable energy). Additionally, this study includes the product's EoL in sustainability considerations and a more environmentally friendly EoL treatment is fostered. This results in the holistic development of a sustainable product. A limitation of this study is the data availability. The ELCD database, which was used for this study, does not provide data for PLA production or EoL recycling. Thus, inventory data from Vink and Davies (2015)were used to model PLA production, and data from Stichnothe and Azapagic (2013) were used to model EoL recycling. Using inventory data from the same source would result in better comparability of results.

Implementing the aforementioned combination of environmental strategies substantially decreased the WPC boards' environmental impacts. For example, climate change results were reduced by 55%. Without integrating ecodesign thinking and a LCA into product development, these reduction potentials remain unused. Thus, the seven-step approach complements the traditional LCA by translating LCA results to quantitative environmental improvement concepts for the entire organization. The majority of environmental impacts are determined during product development (McAloone and Bey 2009). Therefore, integrating ex-ante LCA into the development process contributes to sustainable product development and leads towards maximization of environmental product performance (Buyle et al. 2019).

3.5 Conclusion

The seven-step approach complements conducting an LCA by translating LCA results into environmental improvement efforts. The individual steps help in gaining a better understanding of the product system studied and promotes life cycle thinking. By combining LCA and ecodesign, this approach identifies impact hot spots and derives concepts for environmental improvement. Impact hot spots in the WPC board life cycle are materials (PVC), manufacturing (electricity consumption), and disposal (waste incineration). Environmental concepts derived from these insights propose increasing material and energy efficiency, as well as using less impact-intensive raw materials and energy sources and increased EoL recycling. Implementing these strategies resulted in substantial impact reductions. Because about 80% of a product's environmental impacts are determined during product development (McAloone and Bey 2009), early integration of LCA and ecodesign thinking into product development can identify improvement potentials, which would otherwise remain untapped. This study showed how incorporating ex-ante LCA and ecodesign thinking into the product development process improves a product's environmental performance. The seven-step approach is a suitable tool for this and thus contributes to sustainable product development.

Acknowledgements This work received funding by the Bio Based Industries Joint Undertaking under the European Union's Horizon 2020 research and innovation programme under Grant agreement number 745874 and the Austria Research Promotion Agency (FFG) under the COMET program grant number 865905.

References

Alberta Environment (2001) Acid rain: the science of acidic deposition. Alberta Environment. https://doi.org/10.5962/bhl.title.104506

Allsopp M et al (2001) Incineration and human health: state of knowledge of the impacts of waste incinerators on human health (executive summary). Environ Sci Pollut Res 8(2):141–145. https://doi.org/10.1007/BF02987308

Bianco I et al (2021) Life cycle assessment (LCA)-based tools for the eco-design of wooden furniture. J Clean Prod 324(2021):129249. https://doi.org/10.1016/j.jclepro.2021.129249

Bidoki SM, Wittlinger R (2010) Environmental and economical acceptance of polyvinyl chloride (PVC) coating agents. J Clean Prod 18(3):219–225. https://doi.org/10.1016/j.jclepro.2009.10.006

Boeglin N et al (1997) A feasibility study on boards from wood and plastic waste: bending properties, dimensional stability and recycling of the board. Holz Als Roh- Und Werkstoff. 55(1):13–16. https://doi.org/10.1007/s001070050217

Bovea MD, Pérez-Belis V (2012) A taxonomy of ecodesign tools for integrating environmental requirements into the product design process. J Clean Prod 20(1):61–71. https://doi.org/10.1016/j.jclepro.2011.07.012

Browne M et al (2012) Reducing social and environmental impacts of urban freight transport: a review of some major cities. Procedia Soc Behav Sci 39:19–33. https://doi.org/10.1016/j.sbspro.2012.03.088

Buyle M et al (2019) The future of Ex-Ante LCA? Lessons learned and practical recommendations. Sustainability 11(5456):1–24. https://doi.org/10.3390/su11195456

Clark A, Schroeder J (1977). Biomass of yellow-poplar in natural stands in Western North Carolina. US Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville

Cobut A et al (2015) Reducing the environmental footprint of interior wood doors in non-residential buildings—part 2: ecodesign. J Clean Prod 109:247–259. https://doi.org/10.1016/j.jclepro.2015.05.068

Cordella M et al (2020) Improving material efficiency in the life cycle of products: a review of EU ecolabel criteria. Int J Life Cycle Assess 25(5):921–935. https://doi.org/10.1007/s11367-019-01608-8

Dendromass4Europe (2021) Über Das Projekt. https://www.dendromass4europe.eu/ueber-projekt-post/?lang=de. Accessed 21 July 2021

Deviatkin I et al (2019) Wooden and plastic pallets: a review of life cycle assessment (lca) studies. Sustainability 11(20):5750. https://doi.org/10.3390/su11205750

ELCD 3.2. (2016) European reference life cycle database. GreenDelta GmbH. https://nexus.openlca.org/database/ELCD. Accessed 24 July 2020

Elsom D (1996) Smog alert: managing urban air quality. Earthscan Publ, London

Energochemica (2021) Energo chemica data collection, internal document

European Commission (2010) Mitteilung Der Kommission Europa 2020. Eine Strategie Für Intelligentes, Nachhaltiges Und Integratives Wachstum. European Commission, Brussels

European Commission (2011) Commission staff working paper. Analysis associated with the roadmap to a resource efficient Europe Part I. Accompanying the document communication from the commission to the European Parliament, the council, the European economic and social committee and the committee of regions roadmap to a resource efficient Europe. European Commission, Brussels

European Commission (2015) Proposal for a directive of the European Parliament and of the council amending directive 2008/98/EC on waste. European Commission, Brussels

European Commission (2021) Renewable energy directive 2021. https://ec.europa.eu/energy/topics/renewable-energy/renewable-energy-directive/overview_en. Accessed 4 June 2021

European Committee for Standardization (2014) Composites made from cellulose-based materials and thermoplastics (usually called wood-polymer composites (WPC) or natural fibre composites

(NFC)): part 1: test methods for characterisation of compounds and products. European Standard EN 15534-1. European Committee for Standardization, Brussels
Eurostat (2021) SHARES (Erneuerbare Energien) - Energie - Eurostat. https://ec.europa.eu/eurostat/de/web/energy/data/shares. Accessed 11 May 2021
Gantner J (2012) C-05 transport of wastes to landfill, incineration and recycling facilities—screening and simplified LCA. https://www.eebguide.eu/eebblog/?p=1636. Accessed 2 Feb 2021
González-García S et al (2012) Eco-innovation of a wooden based modular social playground: application of LCA and DfE methodologies. J Clean Prod 27:21–31. https://doi.org/10.1016/j.jclepro.2011.12.028
Groot WJ, Borén T (2010) life cycle assessment of the manufacture of lactide and PLA biopolymers from sugarcane in Thailand. Int J Life Cycle Assess 15(9):970–984. https://doi.org/10.1007/s11367-010-0225-y
Gutiérrez Aguilar C et al (2017) Cleaner production applied in a small furniture industry in Brazil: addressing focused changes in design to reduce waste. Sustainability 9(10):1867. https://doi.org/10.3390/su9101867
Hamer G (2003) Solid waste treatment and disposal: effects on public health and environmental safety. Biotechnol Adv 22(1–2):71–79. https://doi.org/10.1016/j.biotechadv.2003.08.007
Higgins J (2011) Forest industries and the environment. https://www.heritage.nf.ca/articles/economy/forestry-environment.php. Accessed 10 Feb 2021
International Atomic Energy Agency [IAEA] (1999) Health and environmental impacts of electricity generation systems: procedures for comparative assessment. International Atomic Energy Agency, Vienna
Kim JK, Pal K (2011) Recent advances in the processing of wood-plastic composites. Springer, Berlin, Heidelberg
Leadbitter J (2002) PVC and sustainability. Prog Polym Sci 27(10):2197–2226. https://doi.org/10.1016/S0079-6700(02)00038-2
Markarian J (2007) PVC additives—what lies ahead? Plast Addit Compd 9(6):22–25. https://doi.org/10.1016/S1464-391X(07)70153-8
McAloone T, Bey N (2009) Environmental improvement through product development: a guide. Danish Environmental Protection Agency, Copenhagen
McAloone T, Pigosso DCA (2018) Ecodesign implementation and LCA. In: Hauschild MZ, Rosenbaum RK, Olsen SI (eds) Life cycle assessment. Springer, Cham, pp 545–576
Perdomo E et al (2021) Securing sustainable Dendromass production with poplar plantations in European rural areas. D5.5 integration of LCA in value chain establishment: final results. Wood K Plus, Vienna
Petchwattana N et al (2012) Recycling of wood-plastic composites prepared from poly(vinyl chloride) and wood flour. Constr Build Mat 28(1):557–560. https://doi.org/10.1016/j.conbuildmat.2011.08.024
Pigosso D et al (2013) Ecodesign maturity model: a management framework to support ecodesign implementation into manufacturing companies. J Clean Prod 59:160–173. https://doi.org/10.1016/j.jclepro.2013.06.040
Pivato et al (2018) Estimation of global warming emissions in waste incineration and landfilling: an environmental forensic case study. Environ Forensics 19(4):253–264. https://doi.org/10.1080/15275922.2018.1519741
PlasticsEurope (2019) Plastics—the facts 2019. An analysis of European plasticsproduction, demand and waste data. PlasticsEurope, Brussels
Rivela B et al (2006) Life cycle inventory of particleboard: a case study in the wood sector. Int J Life Cycle Assess 11(2):106–113. https://doi.org/10.1065/lca2005.05.206
Rodrigues V et al (2018) Evaluating the potential business benefits of ecodesign implementation: a logic model approach. Sustainability 10(6):2011. https://doi.org/10.3390/su10062011
Sarker M et al (2012) Polyvinyl chloride (PVC) waste plastic treatment using zing oxide (Zno) with activated carbon and produced hydrocarbon fuel for petroleum refinery. Int J Eng Sci 1(8):29–41

Schirp A (2021) Stoffliches WPC-recycling. https://www.wki.fraunhofer.de/de/fachbereiche/hnt/profil/forschungsprojekte/stoffliches-wpc-recycling.html. Accessed 6 July 2021
Sommerhuber PF et al (2015) Substitution potentials of recycled HDPE and wood particles from post-consumer packaging waste in wood-plastic composites. Waste Manage 46:76–85. https://doi.org/10.1016/j.wasman.2015.09.011
Sommerhuber PF et al (2017) Life cycle assessment of wood-plastic composites: analysing alternative materials and identifying an environmental sound end-of-life option. Resour Conserv Recycl 117:235–248. https://doi.org/10.1016/j.resconrec.2016.10.012
Stichnothe H, Azapagic A (2013) Life cycle assessment of recycling PVC window frames. Resour Conserv Recycl 71:40–47. https://doi.org/10.1016/j.resconrec.2012.12.005
Tchertchian N et al (2013) Benefits and limits of a constraint satisfaction problem/life cycle assessment approach for the ecodesign of complex systems: a case applied to a hybrid passenger ferry. J Clean Prod 42:1–18. https://doi.org/10.1016/j.jclepro.2012.10.048
Thornton J (s.a.) Environmental impacts of polyvinyl chloride (PVC) building materials. A Briefing Paper for the Healthy Building Network, s.l
van der Giesen C et al (2020) A critical view on the current application of LCA for new technologies and recommendations for improved practice. J Clean Prod 259:1–12. https://doi.org/10.1016/j.jclepro.2020.120904
van Weenen JC (1995) Towards sustainable product development. J Clean Prod 3(1–2):95–100. https://doi.org/10.1016/0959-6526(95)00062-J
Verband der Deutschen Holzwerkstoffindustrie e.V. (VHI) (2015) Umweltproduktdeklaration Nach ISO 14025 Und EN 15804. WPC Terrassendielen. Institut Bauen und Umwelt e.V., Berlin
Vink ETH, Davies S (2015) Life cycle inventory and impact assessment data for 2014 Ingeo ™ polylactide production. Ind Biotechnol 11(3):167–180. https://doi.org/10.1089/ind.2015.0003

Chapter 4
Background Data Modification in Prospective Life Cycle Assessment and Its Effects on Climate Change and Land Use in the Impact Assessment of Artificial Photosynthesis

Lukas Lazar and Andreas Patyk

Abstract Emerging technologies such as artificial photosynthesis (AP) have the potential to mitigate, avoid or even utilize and remove carbon dioxide by producing chemicals or fuels with sunlight. However, technologies with a very low technology readiness level require a prospective Life Cycle Assessment (LCA) approach to address its uncertainties. Recent developments in LCA software allow for modification of activities across the entire background database and adaptation of results from integrated assessment models. We apply these approaches to a simplified LCA of AP and its fossil-fuel-based-reference product systems. Under the assumptions made, we observe that the application of different scenario approaches to prospective LCA has a major influence on the impact assessment results of the environmental impact categories climate change and land use. The improvement of the emerging technology compared to the reference technology in the climate change impact category may be underestimated by up to 65% if the background database is not adjusted by results from the integrated assessment model. Furthermore, the trade-off between climate change and land use impacts may decrease. For prospective LCA of emerging technologies such as AP, we recommend the inclusion of integrated assessment model results in the LCA background data.

Keywords Prospective life cycle assessment · Background data · Artificial photosynthesis · Photocatalysis · Emerging technologies · Climate change · Land use

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/978-3-031-29294-1_4.

L. Lazar (✉) · A. Patyk
Institute for Technology Assessment and Systems Analysis (ITAS), Karlsruhe Institute of Technology (KIT), Karlstraße 11, 76133 Karlsruhe, Germany
e-mail: lukas.lazar@kit.edu

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*,
Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_4

4.1 Introduction

The IPCC's 2019 special report showed that an additional 0.5 °C of global warming compared to a global temperature increase of 1.5 °C poses additional climate-related risks (IPCC 2019). Nevertheless, according to the latest IPCC report (2021), the 1.5 °C global warming level is likely to be exceeded in all greenhouse gas (GHG) emission scenarios except in SSP1-1.9. This scenario assumes "very low and low GHG emissions and CO_2 emissions declining to net zero around or after 2050, followed by varying levels of net negative CO_2 emissions" (IPCC 2021, p. SPM-15). This means that even if GHG emissions are reduced immediately, the removal of CO_2 from the atmosphere will most likely be necessary to keep global warming below the 1.5 °C level. Besides CO_2 removal with biomass, carbon capture and utilization are conceivable technological solutions for large-scale CO_2 removal. Carbon capture and storage may involve trade-offs in terms of environmental and human health impacts due to direct and indirect impacts of increased fuel consumption, additional processes and materials and side effects on land, water and soil (Schreiber et al. 2009, 2010; Terlouw et al. 2021a; Volkart et al. 2013). Carbon capture and utilization, on the other hand, offers the potential to produce carbon-based products independent of fossil resources such as crude oil and natural gas while simultaneously removing[1] or recycling CO_2.

One possible carbon capture and utilization technology is artificial photosynthesis[2] (AP), also referred to as photocatalysis. Derived from natural photosynthesis, it mimics the idea of using captured sunlight to convert, e.g., water and carbon dioxide into basic chemicals, fuels, or intermediates. AP dates back to 1839, when the photovoltaic effect was first observed by Becquerel (Wenham 2009). In the early 1970s, Fujishima and Honda (1972) described the photolysis of water. Unlike natural photosynthesis, whose estimated efficiency is mostly limited to 1%[3] (Barber and Tran 2013; Blankenship et al. 2011), AP aims for efficiencies of more than 10% (Bonke et al. 2015; Hoffmann et al. 2011; Kim et al. 2022; Larkum 2010; Tahir and Amin 2013). Moreover, it offers the potential for a wide range of final products as recombinations from the carbon- and hydrogen-based input materials, e.g. synthesis gas (syngas), oxygenates, aldehydes, and acids. However, the photocatalytic reduction of CO_2 in particular presents an additional challenge due to the high stability of CO_2 (Seo et al. 2017). To verify and assess the potential environmental benefits of carbon capture and utilization with AP, including the upstream and downstream processes, Life Cycle Assessment (LCA) is applicable. LCA provides a systematic method for identifying the environmental impacts of products or services (European Commission 2010, 2011; International Standard Organization 2006a, b).

[1] Net CO_2 removal only applies to long-lasting products in which CO_2 is stored in the long term.

[2] Artificial photosynthesis as a generic term includes photocatalytic water splitting and carbon dioxide reduction but also biomimetic systems and others, which we do not consider in this study.

[3] The estimated maximum efficiency of photosynthesis is about 4.5% and is only reached under optimum conditions and normally does not exceed 1–2%. The approximate efficiency of global photosynthesis is around 0.2% (Barber and Tran 2013).

Studies that have investigated AP product systems using LCA approaches have shown that solar-to-chemical-energy conversion efficiency plays a major role in the impact assessment: To achieve the environmental performance of the reference product system, efficiencies of more than 2–5% were necessary, depending on the impact categories used and the products considered (Dincer and Acar 2015; Dufour et al. 2012; Patyk et al. 2020; Sathre et al. 2016; Trudewind et al. 2014; Zhai et al. 2013). Moreover, according to Sathre et al. (2016), a cell lifetime of at least five years is required to achieve a positive net energy value. Most of the identified literature focused on the cumulative energy demand and the climate change impact category.

Because LCA was originally developed to assess existing technologies, technology assessments typically assume the conditions of a present techno- and biosphere without considering differences caused by past and/or future elementary flows and characterization factors. This can lead to unpredictable uncertainties when assessing emerging technologies with a low technology readiness level (TRL) such as AP, which will most probably not be applicable in the near future. Consequently, the material and energy flows on an industrial scale can only be modeled to a certain extent and therefore must be estimated. However, this can lead to high uncertainties. Parvatker and Eckelman (2019) evaluated Life Cycle Inventory (LCI) generation methods and found that stoichiometric calculations underestimate actual global warming results by 35–50%. Furthermore, the technologies will be introduced in the future rather than today. Hence, production and operation may change and the future technosphere will most likely influence the results. To cope with these challenges, prospective LCA has been proposed where future scenarios are used to create probable future solution spaces and/or discussing the scaling of emerging technologies to an industrial scale (Cox et al. 2018; Ioannou et al. 2021; Maes et al. 2021; Marini and Blanc 2014; Mendoza et al. 2020; Parisi et al. 2020; Pizzol et al. 2021; Terlouw et al. 2021b; Tsoy et al. 2020). In addition ready-to-use open source tools were provided (Joyce and Björklund 2021; Sacchi 2020; Sacchi et al. 2022) as well as recommendations and lessons learned (Bergerson et al. 2020). Furthermore, Thonemann et al. (2020) reviewed studies on prospective LCA and developed a methodological framework for its application that addresses the goal and scope definition, the LCI data generation and modification, the use of different characterization factors in the LCIA step, and the uncertainty analysis and communication in the interpretation step.

Besides using methods for scaling and dealing with uncertainties in the case of emerging technologies, prospective LCA can use different intensity levels for future solution spaces. The most straightforward approach is to implement future changes in the foreground model, such as future energy mixes and/or efficiency gains. For example, process energy, heat, and steam are modified in the modeled foreground product system, but energy supply for upstream processes, including the production of equipment, facilities, and infrastructure, is assumed to remain unchanged. However, particularly for renewable energy systems based on dissipated energy, the background model could result in higher material and land use. Recent developments allow for automated modification of activities across the entire database (Mutel and Cox 2021), making it possible to extend the previous modification of foreground data to the entire background data. Furthermore, the open source tool *premise*[4] developed by Sacchi et al. (2022) enables the use of integrated assessment model results and their integration into LCA databases. These scenario implementations include predicted electricity mixes, add new technologies, and adjust efficiencies.

Since modifications to the foreground model do not take into account possible changes in the background, there is a potential risk that the results of the impact assessment and the interpretation of the results are subject to additional uncertainties. In particular, for product systems whose main impacts are in the background, there could be a bias regarding their environmental footprint. Moreover, impacts in the reference product systems could be over- or underestimated, because potential reductions in climate change impacts or increases in land use impacts from future renewable energy mixes in the background are not considered. Mendoza et al. (2020) proposed and tested changing the electricity background processes in a prospective LCA based on scenario results from an integrated assessment model. When comparing electric and internal combustion engine vehicles, they found that market changes in the electricity background processes resulted in a 14–80% reduction in climate change impacts. The deviation could be even higher for renewable energies, that depend on distributed energy sources such as AP and thus mainly on the background processes. This raises the research question of whether background data have a major effect on the impact assessment of AP and how sensitive these results are to the corresponding background data modification method and scenario used.

To analyze and quantify these effects, we aim to compare the AP impact assessment results obtained from different foreground and background data modification approaches and different scenarios within them. We focus on AP with different products, as an example of an emerging technology. To this end, we develop and implement a simplified theoretical model of AP that provides the Life Cycle Inventory (LCI) data and enables the data modification procedure. Beyond climate change, the implementation of renewable energies might lead to trade-offs, in particular between the impact categories climate change, land use and mineral and metal resources (Fuss and Xu 2021; Laurent et al. 2018; Vidal et al. 2013). We therefore added the impact category land use to the investigation to analyze one of these trade-offs. The contributions of this paper can be summarized as follows:

- We present simplified preliminary LCI data for the emerging technology AP producing oxalic acid, glyoxal, formic acid, acetic acid and syngas.
- A Python script is provided for the exchange of energy supply processes (power, heat, steam, hydrogen) in the inventory database.
- Life Cycle Impact Assessment (LCIA) results of AP with modified foreground and background data are generated; deviations, trends, and effects are analyzed.

Section 4.2 describes Goal and Scope and LCI of the LCA study as well as the methods used to generate the prospective LCI of AP and to realize the background

[4] *Premise* allows to modify the inventory database ecoinvent by the integration of results from integrated assessment models such as REMIND from the Potsdam Institute for Climate Impact Research (PIK). The code can be accessed at https://github.com/polca/premise.

data exchange. Next, the LCIA is presented in the results (Sect. 4.3), including the contribution, sensitivity, and scenario analysis. The results are discussed in Sect. 4.4, conclusions and recommendations are given in Sect. 4.5 along with implications for future research.

4.2 Methods

4.2.1 Life Cycle Assessment: Goal and Scope

The LCA used in this study aims to quantify and compare the cradle-to-gate environmental performance of the functional unit, defined as the production of one kilogram of oxalic acid, glyoxal, formic acid, acetic acid, or syngas at 25 °C and atmospheric pressure, either by the emerging AP technology or by a reference product system. The emerging technology assessed in this study corresponds to a TRL of 2–3 and is therefore subject to high uncertainties. The reference product system is modeled with data from the ecoinvent inventory database and results from process simulations representing mature technologies (RWTH Aachen 2014; Wernet et al. 2016). Results from a previous research project were used to identify potentially beneficial products and pathways compared based on their cumulative energy demand (Patyk et al. 2020). Conventional technologies with the same primary function were modeled as reference products. When assessing an emerging technology, recent findings from ex-ante or prospective LCA were considered in terms of LCI data generation methods and background data modification. Several scenarios were built, partly with complete modifications of the database and coupling with integrated assessment model results. This allows assessing both the new technology and the reference technology in a prospective future setting. The product system of AP consists of the main process, a photocatalysis and the upstream supply chain for water, methane and carbon dioxide (Fig. 4.1). The photocatalysis comprises the photoreactor including the photocatalyst and the corresponding manufacturing processes.

In the background data, economic allocation according to the ecoinvent standards is used together with the ecoinvent cut-off system model for burdens and credits of recycling (Weidema et al. 2013). Consequently, we applied economic allocation to the foreground modeling and assessed the product systems from an attributional perspective. For the LCA and LCIA, we used the open source software tools brightway2 (Mutel 2017) and the Activity Browser (Steubing et al. 2020) together with ecoinvent 3.7.1 (Wernet et al. 2016) and ILCD midpoint 2.0 (“climate change, total” and “resources, land use”) impact assessment. This LCA is limited due to the early stage of development of the assessed technology and thus represents a first estimate of the environmental performance in a best-case scenario.

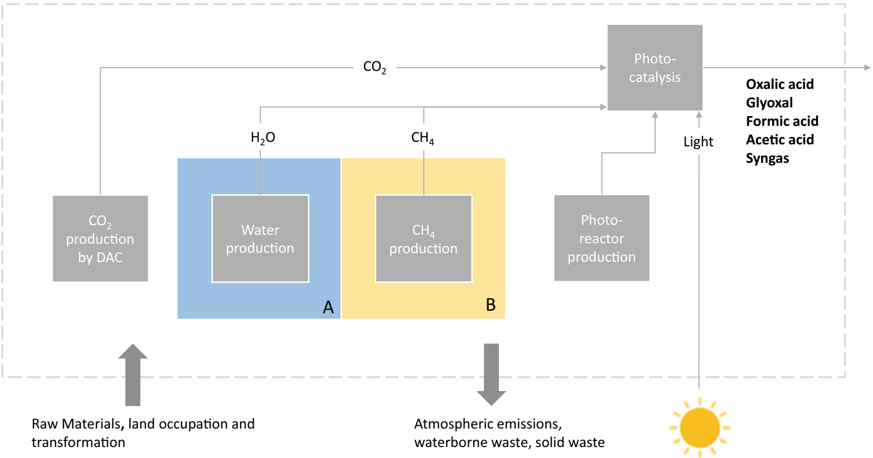


Fig. 4.1 Schematic diagram of the artificial photosynthesis (AP) product system with flows, processes, and system boundaries. Upstream and downstream processes, materials, services and emission flows are included in the depicted process

4.2.2 Prospective Life Cycle Inventory of Artificial Photosynthesis

Creating a LCI for an emerging technology with a TRL of approximately 2–3 requires the use of LCI generation methods because no plant or industry data is available. According to the hierarchy of methods developed by Parvatker and Eckelman (2019), we proceeded as follows:

- Process simulation data for the reference product system in the case of syngas.
- Stoichiometry to estimate the input and output mass flows of the considered AP product systems.
- Basic process calculations to determine the solar energy input and estimate the energy demand for production of the photocatalyst proxies.
- Proxy processes for the photoreactor, the photocatalyst, and the reference product system of oxalic acid.
- The step for the production of the photocatalyst material tungsten trioxide from ammonium paratungstate was omitted.

Simplified energy balance of AP. To build an inventory of AP using sunlight by determining the solar energy input and consequently the size of the photoreactor, we use the solar-to-hydrogen (STH) efficiency, which, despite its name, can be defined more generally as chemical energy produced divided by solar energy input (Chen et al. 2010; Zhai et al. 2013):

$$\mathrm{STH} = \frac{\text{amount of substance}\left[\frac{\mathrm{mmol}}{\mathrm{s}}\right] \cdot \text{Gibbs free energy}\left[\frac{\mathrm{kJ}}{\mathrm{mol}}\right]}{\text{solar energy input}\left[\frac{\mathrm{mW}}{\mathrm{cm}^2}\right] \cdot \text{area}\left[\mathrm{cm}^2\right]} \quad (4.1)$$

Instead of measuring a mass flow and calculating the STH, we aim to calculate the solar energy input using a predefined estimate for the STH. Reformulation of Eq. 4.1 returns the solar energy input for one mole of product multiplied by the area. Consequently, Gibbs free energy is calculated using the standard enthalpy of formation and the standard molar entropy:

$$\text{Gibbs free energy} = \Delta \mathrm{H}^0 - \mathrm{T} \cdot \Delta \mathrm{S}^0 \tag{4.2}$$

For the standard enthalpy H^0 and entropy S^0, we use data from the NIST Chemistry WebBook (National Institute of Standards and Technology 2021) and the Computational Chemistry Comparison and Benchmark DataBase (National Institute of Standards and Technology 2020).

4.2.3 Scenario Building in Prospective LCA

We use three different scenario approaches with increasing depth of detail: Foreground data modification (FG), complete background database customization (BG), and database construction based on the results of an integrated assessment model (IAM). Figure 4.2 shows the tools used and the scenarios built as well as the coupling with the LCA structure. The foreground and background modification approaches implement the electricity mix, projected by Pfluger et al. (2017), in the 2030 and 2050 scenarios and heat from electricity in the 2050 and Run-of-river scenarios. The Run-of-river scenario uses only electricity from run-of-river plants and represents the energy technology with the lowest environmental impacts available in the ecoinvent database for Germany (Wernet et al. 2016). In the background modification approach, the LCI tool *wurst*,[5] developed by Mutel and Cox (2021), is used to substitute the processes in the entire database. The corresponding Python script is provided in the electronic supplementary material. In the IAM approach, the following predefined scenarios from the integrated assessment model REMIND are applied (Baumstark et al. 2021; Luderer et al. 2013):

- the Base scenario is market-based and does not consider any specific climate policies, leading to global warming of more than 3.5 °C by 2100;
- the National Determined Contributions (NDC) scenario assumes that all emission reductions and other mitigation commitments under the Paris Agreement are implemented;
- the National Policies implemented (NPi) scenario projects that policies fail to achieve NDC targets in 2030 and follow currently implemented policies;

[5] *Wurst* is a Python package, which allows modification and exchange of activities in LCI databases. The code can be downloaded from the following website: https://github.com/polca/wurst.

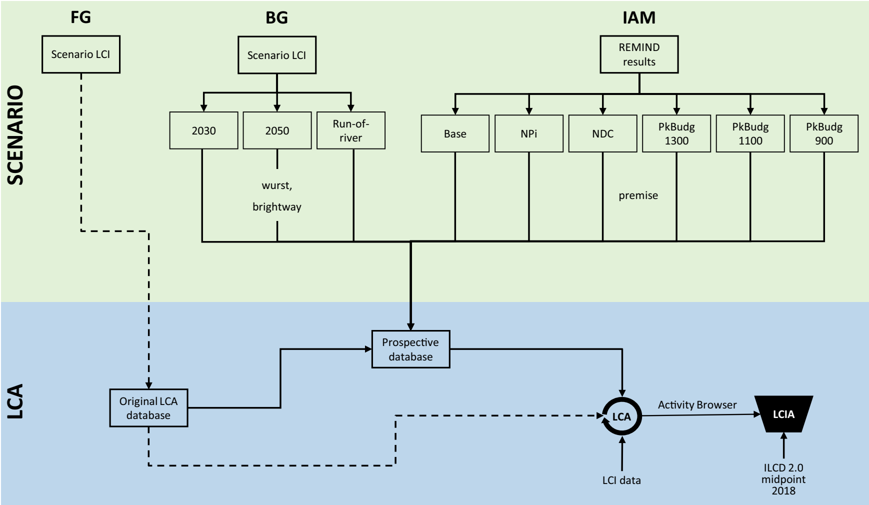


Fig. 4.2 Simplified diagram of the scenario building methodology and the coupling with the LCA scheme. While the foreground (FG) approach is directly connected to the original LCA database, the background (BG) approach first modifies the database using the tool *wurst*. Integrated assessment model results are implemented in the IAM pathway using the *premise* tool. The prospective inventory databases are used together with the Life Cycle Inventory (LCI) data from the artificial photosynthesis (AP) model and the reference product system for the Life Cycle Impact Assessment (LCIA)

- the PkBudg (900/1100/1300) scenarios limit the cumulative release of greenhouse gas emissions to either 1300 Gt (below 2 °C global warming), 1100 Gt (well below 2 °C), or 900 Gt (below 1.5 °C).

Based on the output of the REMIND scenarios, energy-intensive sectors (power generation, cement, steel production, freight and passenger road transport, supply of conventional and alternative fuels) are adjusted in the ecoinvent database and efficiencies are updated (Sacchi et al. 2022). More details about modifications by the *premise* tool are described in its documentation (Paul Scherrer Institut and Potsdam Institute for Climate Impact Research 2022).

4.2.4 Life Cycle Inventory

The LCI was created assuming a photocatalysis based on theoretical considerations, currently with a TRL of 2–3. Due to its early stage of development, the LCI of the photocatalysis is not a complete description of a working technology, as usually common in LCA, but rather a first estimate of likely materials and energy flows. The detailed LCI information is provided in the electronic supplementary material (Table SM 4.1).

Table 4.1 Chemical formula and Gibbs free energy of the considered products (National Institute of Standards and Technology 2020, 2021; Patyk et al. 2020)

Product name	Chemical formula	Gibbs free energy (kJ)
Acetic acid	CH_3COOH	872
Formic acid	$HCOOH$	544
Glyoxal	$C_2H_2O_4$	1655
Oxalic acid	$C_2H_2O_4$	637
Syngas	$H_2 + CO$	171

Photocatalysis. The photocatalysis is the core part of the AP product system and requires a photoreactor including the photocatalyst. For the solar-to-chemical-energy efficiency, we assume 5% as a first estimate, in line with the literature, additionally taking into consideration that the assumed ideal product pathway will not be achievable (Dincer and Acar 2015; Dufour et al. 2012; Patyk et al. 2020; Sathre et al. 2014, 2016; Trudewind et al. 2014; Zhai et al. 2013). The solar energy input is generated using the Photovoltaic Geographical Information System (PVGIS) for the location of Karlsruhe in Germany (European Commission 2021). We calculated the P90 value of the direct solar radiation using data for the years 2005–2016, resulting in 1223 kWh/m^2 per year. The P90 value represents a conservative estimate used in investment decision making for concentrated solar power projects (Telsnig 2015). As potential products, we selected chemicals with the lowest break-even point of their cumulative energy demand compared to their reference product systems determined by the research project PROPHECY (Patyk et al. 2020). In addition, we added syngas production by photocatalytic dry reforming of methane to include a potential synthetic fuel route in the assessment. Syngas is one of the most important raw materials for the production of value-added industrial chemicals and synthetic fuels such as methanol, dimethyl ether, and Fischer–Tropsch diesel (Schakel et al. 2016; Yang et al. 2021). We calculated the Gibbs free energy (Table 4.1) for the considered products and reactions according to Eq. 4.2 in Sect. 4.2.2.

Photoreactor. Compared to a chemical reactor, the photoreactor must ensure that the photons from the light source and the reactants come into contact with the photocatalyst (Braham and Harris 2009). Particularly in the field of photocatalytic pollutant treatment, several prototypes have been reported by Lasa et al. (2005). Moreover, reactor designs were reviewed by Tahir and Amin (2013) for CO_2 reduction on a laboratory scale and discussed by Braham and Harris (2009) for solar photocatalysis. Braham and Harris (2009) found compound parabolic collectors to be the most suitable for photocatalytic applications. However, the only suitable collectors with available LCI data in ecoinvent were parabolic trough collectors (“concentrated solar power plant construction, solar thermal parabolic trough, 50 MW”). For the application as a photocatalytic reactor, the catalyst is envisaged to be applied to a reactor tube, so that the available LCI data could be used with minor modifications. Advantages of parabolic trough collectors are the smaller amount of photocatalyst necessary because of its ability to concentrate light and the possibility of higher operating temperatures and pressures due to a smaller absorber diameter of the absorber. However, these advantages are offset by the disadvantages that only direct sunlight can be used and that a tracking mechanism, at best a dual-axis tracker, is required. The original ecoinvent activity was modified from the application of electricity production to photocatalysis; unsuitable materials and subprocesses were removed (SI Table 1). The lifetime of the photoreactor was assumed to be 20 years.

Photocatalyst. Materials from photoelectrochemical water-splitting electrodes were used as a proxy, since no detailed LCI data were available for the application considered. Zhai et al. (2013) calculated 4.7 g/m^2 silicon (Si) for the photocathode, 0.03 g/m^2 platinum (Pt) as catalyst, and 0.1 g/m^2 tungsten trioxide (WO_3) for the photoanode as a hypothetical design for water splitting. Tungsten trioxide is not available in ecoinvent, but ammonium paratungstate is. Therefore we calculated the amount needed to produce tungsten trioxide stoichiometrically, according to Lassner and Schubert (1999). Heat, electricity, and the chemical factory for the final fabrication step of tungsten trioxide from ammonium paratungstate are neglected. For the fabrication process of the photoelectrodes and catalysts, Zhai et al. (2013) calculated the primary energy demand for heating, vacuum pumping, and plasma generation based on the first law of thermodynamics, assuming a thermal efficiency of 50% (medium case 1410 MJ/m^2). We recalculated the electricity-to-primary energy conversion and directly used electricity for the LCI data, representing the fabrication of the photocatalyst. We assumed a lifetime of 10 years for the photocatalyst.

Carbon dioxide, methane, and water supply. To obtain the targeted products by the photocatalysis, the reactants carbon dioxide and water or methane (in the case of syngas) are required. For the supply of carbon dioxide, we assume a Direct Air Capture (DAC) plant, which is based on data from Terlouw et al. (2021b). These data are confirmed by the DAC manufacturer. However since it was not possible to publish the original data, they simplified the LCI, leading to similar results. In the case of the syngas route (B), we apply "biogas purification to biomethane by pressure swing adsorption [CH]". Since biogas purification or upgrading produces biomethane and carbon dioxide, both products were used for the syngas pathway and only the missing carbon dioxide was supplied by DAC. For the water supply, we use the ecoinvent process "market for water, deionised [Europe without Switzerland]".

Energy and integrated assessment model scenarios. For the FG and BG scenario approaches, we used projected electricity mixes for Germany in the years 2030 and 2050 from Pfluger et al. (2017). We used their base scenario, which considers the achievements of energy and climate policies at minimum costs. This scenario targets electricity production excluding imports and exports of 33% wind, 22% other renewables, and 45% fossil power in 2030 and 67% wind, 21% other renewable, and 12% fossil power in 2050. Pfluger et al. (2017) use categorizations of electricity suppliers different from those used by ecoinvent, therefore an allocation was required. Electricity imports and exports, future efficiency changes and lower impacts were neglected. The full LCI is provided in the electronic supplementary material (Table SM 4.1). For heat from electricity, considered in the 2050 and Run-of-river scenarios, we assumed that electricity is converted into heat with 100% efficiency. The LCI data for a heating infrastructure was not considered. In addition, in the Run-of-river scenario, hydrogen production was substituted by hydrogen based on electrolysis ("chlor-alkali electrolysis, membrane cell [RER])." For the

IAM approach, we used the *premise* tool's implementation of the REMIND model. *Premise* updates the electricity mixes, adds new technologies, and adjusts efficiencies in electricity, vehicle, cement, steel, and solar photovoltaic processes (Sacchi et al. 2022). For more details on the REMIND scenarios, see Paul Scherrer Institut and Potsdam Institute for Climate Impact Research (2022) and on the model itself, see Baumstark et al. (2021) and Luderer et al. (2013).

Reference product systems. For the reference product systems of formic acid (methyl formate route), acetic acid (without water, in 98% solution state), and glyoxal, we used ecoinvent data for Europe (RER) (Wernet et al. 2016). We preferred the most common routes and avoided market processes because impacts from transport and storage are not considered for AP routes either. For oxalic acid, no conventional route in the form of inventory data was available. Vahidi and Zhao (2017) used acetic acid instead of oxalic acid since both can be produced from carbon monoxide on an industrial scale. They also referred to Nuss and Eckelman (2014), who used the same substitution in their LCA. In the absence of alternatives, we used acetic acid as a proxy for the reference product system of oxalic acid knowing that high uncertainties might occur. Furthermore, for the reference product system of syngas (H_2:CO = 1) production, we used data from the CO2RRECT project (RWTH Aachen 2014; Sternberg and Bardow 2015), which is based on a process simulation by Bayer Technology Service. We changed the H_2:CO ratio by economic allocation of the surplus hydrogen; however we did not consider a separation process which can further increase the impacts. To include the production facility, we scaled the ecoinvent process "chemical factory construction [RER]". In addition, we compared climate change impacts with data from studies using different sources for syngas production (Artz et al. 2018; Sternberg and Bardow 2016). The variance of 5–20% is reasonable taking into account the unequal H_2:CO ratios, the different LCI databases and impact assessment methods used.

4.3 Results

4.3.1 Life Cycle Impact Assessment

The impact categories climate change and land use were investigated using three prospective LCA approaches and in a total of 18 scenarios for the years 2030 and 2050. In the climate change impact category, the AP product system performs favorably compared to the reference product system (REF) (Fig. 4.3). The AP pathways achieve potential negative cradle-to-gate CO_2-equivalents in all scenarios considered. The negative climate change impacts stem from the assumption that CO_2 is captured and used (Fig. 4.4). The main contributors to climate change and land use impacts stem from DAC and the photoreactor production. The contribution of the photocatalyst is very small (<1%).[6] The overall trend of the scenarios for both the AP and the REF product systems indicates a downward trend in the more future and optimistic scenarios, particularly in the BG-Run-of-river and IAM-2050PkBudg900 scenarios (Fig. 4.3). Scenarios with background data modification have in most cases lower impacts compared to the foreground modification scenarios. The BG/IAM scenario approach decreases the climate change impacts of AP by up to 60/64% (acetic acid), 47/49% (formic acid), 58/60% (glyoxal), 41/42% (oxalic acid) and 82/41% (syngas). Concurrently, the impacts of REF decrease by up to 60/44% (acetic acid), 51/23% (formic acid), 40/31% (glyoxal), 60/44% (oxalic acid) and 6/6% (syngas). An exception is the BG-2030 scenario where the impacts are higher than in the FG-2030 scenario. This is due to the projected German electricity mix in 2030, which emits 473 g CO_2-eq/kWh compared to 413 g CO_2-eq/kWh in the European power mix,[7] resulting in an increase if used for the entire database.

In the impact category land use, the impacts of the AP product systems exceed the impacts of the REF product systems in all scenarios considered, which is due to the high land use by the solar collectors, which were assumed to serve as photoreactors. However, in the IAM-2050 scenarios, land use slightly decreases in both product systems compared to IAM-2030. Thus, the trade-off between climate change impact reduction and increase in land use impacts can be reduced in the IAM scenarios compared to the FG scenarios, with a 7–80%[8] reduction in land use impacts measured. BG-Run-of-river remains the most favorable scenario also in terms of land use. However, this scenario is an outlier due to the very beneficial performance of run-of-river power and the assumed heat supply from electricity. In the land use impact category, the impacts of the background modification in 2030 (BG-2030) exceed the impacts of the foreground modification in 2030 (FG-2030). The projected electricity mix which we modeled for Germany in 2030 shows a slightly higher land use of 2.8 compared to 2.5 points per kilowatt hour in the European market mix. The contributions increasing land use come from photovoltaics, hard coal, and infrastructure.

The difference in performance of the AP system compared to the REF product system varies across scenario approaches, scenarios, and products for the two impact categories climate change and land use (Fig. 4.5). In the climate change impact category, the AP system performs favorably in the production of acetic acid with an improvement of 145–313%, oxalic acid 139–261%, glyoxal 127–194%, syngas 109–188%, and formic acid 111–138%. In the land use impact category, the AP system performs worse than REF in the production of glyoxal 319–1817%, oxalic acid 49–333%, formic acid 9–602%, acetic acid 392–1665%, and syngas 96–2364%. The overall variance in both impact categories is lowest in the FG scenarios (except

[6] Measured in the FG-2050 scenario. The contribution might be slightly higher in other scenarios.

[7] Referring to the ecoinvent 3.7.1 process: “market group for electricity, medium voltage [RER].”

[8] Measured from the minimum of the relative change (AP to REF) in the FG scenarios to the minimum of the relative change in the IAM scenarios. The same procedure is applied to the maximum of FG and IAM.

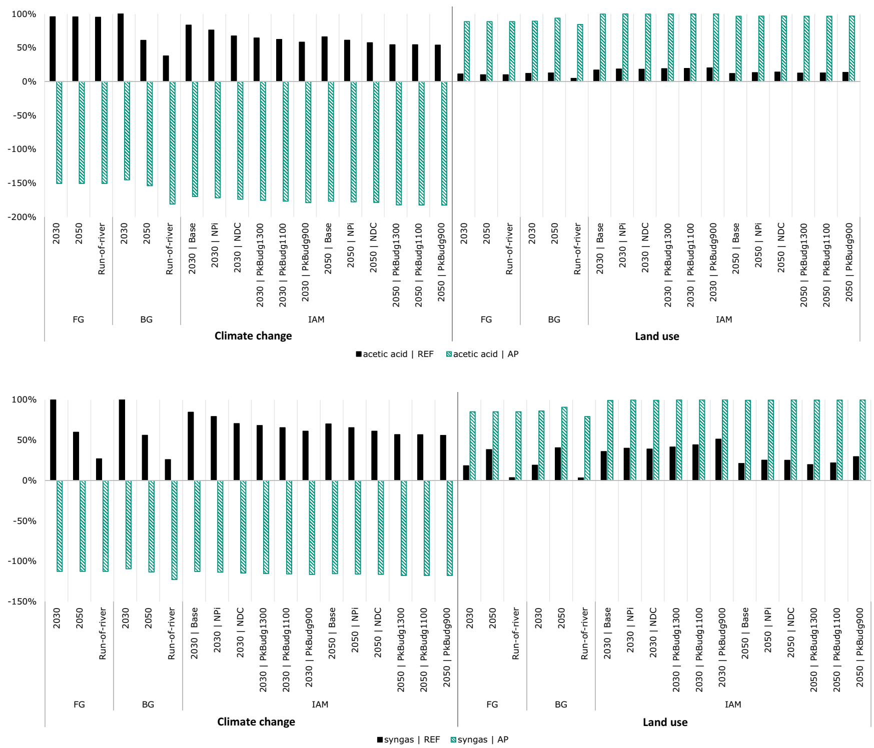


Fig. 4.3 Life cycle impact assessment results for acetic acid and syngas production by artificial photosynthesis (AP) and its fossil reference product system (REF) considering foreground (FG), background (BG), and integrated assessment model (IAM) scenario approaches with the respective scenarios FG and BG: 2030, 2050, Run-of-river; IAM: market-based (Base), National Policies implemented (NPi), National Determined Contributions (NDC), emission budgets (PkBudg900/1100/1300). Results are normalized to the highest climate change and land use impacts. Absolute values with results for formic acid, glyoxal and oxalic acid are provided in the electronic supplementary material

for formic acid and syngas), high in the BG scenarios and medium to high in the IAM scenarios. Scenarios using background data modification allow a higher potential reductions in climate change and land use impacts in most of the cases. Hence, comparing the FG and IAM scenario approaches, the climate change impact improvement could be underestimated[9] by up to 65% (acetic acid), 47% (oxalic acid), 33% (glyoxal), 10% (formic acid), and 2% (syngas). For land use, the results could be overestimated by up to 83% (syngas), 80% (formic acid), 54% (oxalic acid), 47% (glyoxal) and 43% (acetic acid). However, in their maxima, the scenarios, especially

[9] Measured from the minimum of the relative change (AP to REF) in the FG scenarios to the minimum of the relative change in the IAM scenarios. The same procedure is applied to the maxima of FG and IAM.

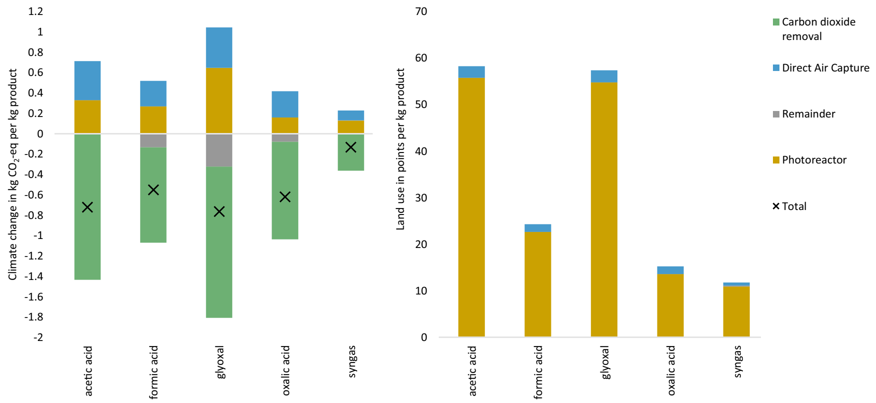


Fig. 4.4 Contribution to the impact categories climate change (left) and land use (right) in the FG-2050 scenario showing the main contributors and the total impacts when negative climate change impacts are included

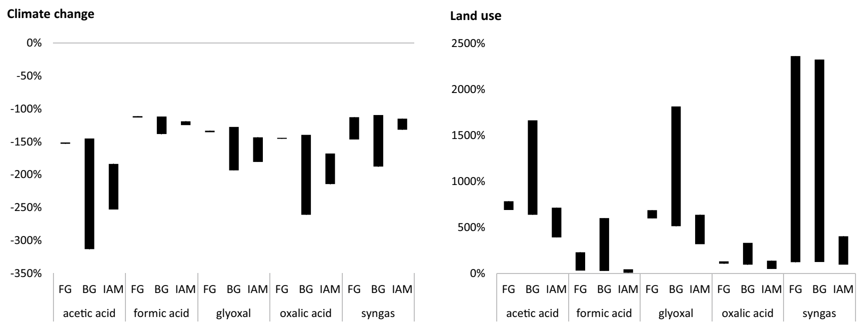


Fig. 4.5 Artificial photosynthesis (AP) performance normalized to its reference product system, including the result variance of the scenarios in the foreground (FG), background (BG), and integrated assessment model (IAM) scenario approaches

BG, can lead to a very high increase in land use impacts, worsening the performance of AP by up to 2364% compared to REF.

The impact assessment results, both in the climate change and land use impact categories, follow a trend consistent with the scenarios. A correlation analysis[10] showed that the correlation for climate change and land use impacts in the scenarios is high between most of the products, with the lowest value being between AP oxalic acid and REF syngas (r = 0.25; r = 0.67) and a *p*-value lower than 1×10^{-4}. The mean of all correlation coefficients in the results for climate change impacts is 0.80 and for land use impacts 0.88. If the correlation were sufficiently high, it could be

[10] The correlation coefficient (spearman) and the p-value were calculated with the Python library SciPy version 1.6.3.

possible to predict the impacts of a single scenario to save time and computational resources. The complete results of the correlation analysis, including the regression equations, are provided in the electronic supplementary material (Table SM 4.2).

Sensitivity analysis. The parameters used for the solar-to-chemical energy efficiency, the photocatalyst, and the photoreactor were estimated or based on proxy processes. Since, the photoreactor, in particular, contributes significantly to the LCIA results, these parameters were modified to analyze the sensitivity of the model. The solar-to-chemical energy efficiency was varied in 1% increments from 1 to 20%, and the photocatalyst and photoreactor were scaled in 0.1 increments up to a maximum of 10. The variation was applied to the FG-2050 scenario in the case of acetic acid.

The results of the sensitivity analysis of acetic acid in the FG-2050 scenario show that the model is very sensitive to a variation in solar-to-chemical energy efficiency. Therefore, if the input parameter of 5% cannot be achieved, the climate change and land use impacts can change significantly. Particularly in the 1–5% range, climate change and land use impacts decrease by 221 and 79%, respectively. Scaling up the photoreactor by a factor of two increases the climate change and land use impacts by 46 and 96%, respectively. In contrast, photocatalyst production shows comparatively low variance: A scaling factor of 10 increases climate change impacts by less than 10% and the land use impacts by less than 1% (Fig. 4.6).

4.4 Discussion

The results of this study, based on the investigation of the scenario approaches in the LCA of AP, are subject to limitations imposed by the methodology used and the assumptions made, particularly due to the very low TRL of AP. We modeled the AP product system with assumed reactions and efficiencies that are not available today. Ideal conversion was assumed, losses, e.g., from side reactions were not considered. The inventory was based neither on an industrial process nor on a prototype. The applied LCI methods proposed by Parvatker and Eckelman (2019) are in the mid and lower range of the hierarchy, meaning that the accuracy can be low. However, higher ranged methods such as plant data are not yet available for the assessed technology. An LCA at this early stage in the development of a technology inherently has drawbacks regarding the certainty of the results. Parvatker and Eckelman (2019) state that stoichiometric methods should only be used when no process configuration or plant data are available, which is the case for AP. In their results, they show that stoichiometric calculations can underestimate the climate impacts and either under- or overestimate the land use impacts.

The assumed solar-to-chemical-energy efficiency of 5% was not confirmed in a prototype. Efficiencies achieved, e.g., in photoelectrocatalytic CO_2-reduction, range from 0.97% for solar-to-fuel conversion to 1.28% for solar-to-acetate conversion and are thus much lower (Tahir and Amin 2013). Nevertheless, May and Rehfeld (2019) calculated that, ideally, 19% of the incident solar photons convert CO_2 into a storable product. Furthermore, the applied reaction mechanism and the calculated

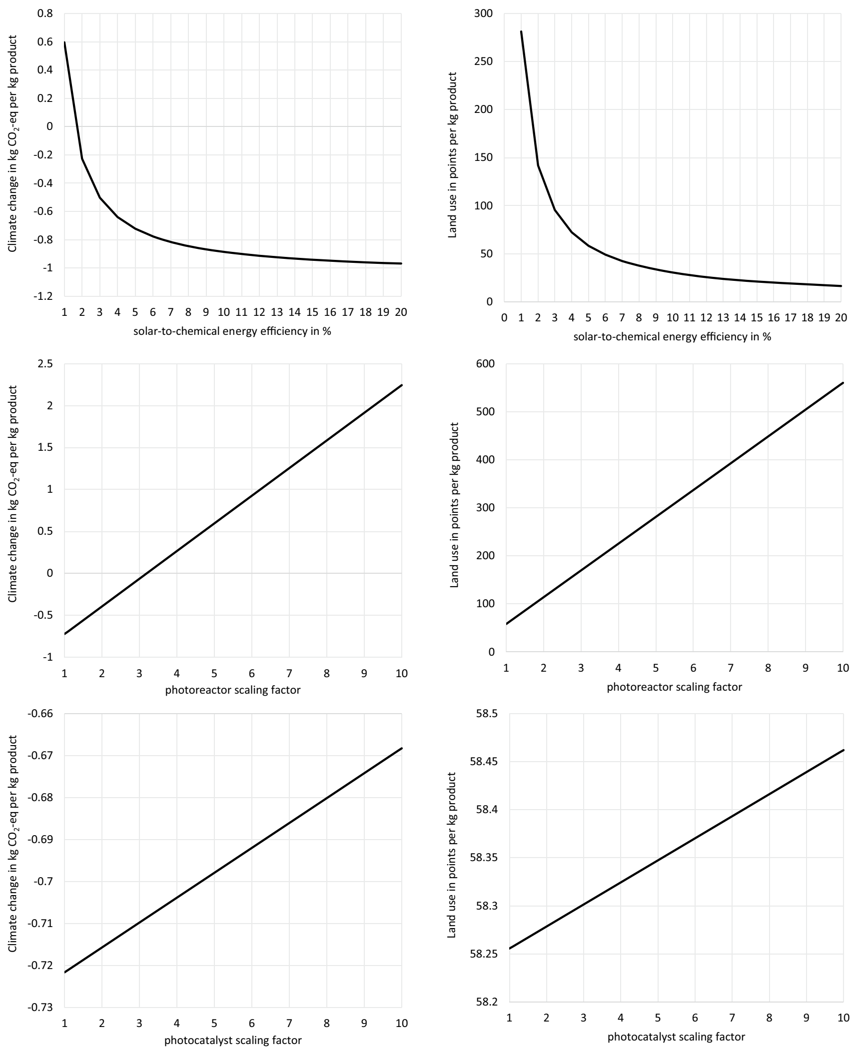


Fig. 4.6 Sensitivity of climate change (left) and land use (right) impacts to solar-to-chemical energy efficiency parameters and scaling factors for the photocatalyst and photoreactor production. The results show the artificial photosynthesis (AP) product system of acetic acid in the FG-2050 scenario

Gibbs free energy do not take into account the constraints related to the potentials of the conduction and valence band. These potentials determine the possibility of producing a particular product, and therefore the production pathway shown may be impossible in a practical application (Karamian and Sharifnia 2016). Due to lack of data, we used materials and manufacturing efforts for the catalysts from the literature based on a thermodynamic model for use in hydrogen production. Therefore, we apply catalyst materials of photoelectrodes from a photoelectrochemical cell to a photocatalytic reactor. Pipes, valves, and pumps were considered for another application (thermal oil) as already implemented in the dataset and not adjusted for AP application. Decommissioning was not considered, and recycling is only accounted for in the input materials according to the ecoinvent methodology (Weidema et al. 2013). Due to data availability, parabolic trough collectors were used as photoreactors, even though a compound parabolic collector might be more suitable for AP (Braham and Harris 2009). However, the assessment provides a first rough estimate of the baseline environmental performance in a best-case scenario. If the product system already performs unfavorably compared to the reference product systems, it is very likely that environmental impacts will further increase with a more complex assessment.

The prospective LCA scenarios were based on different assumptions and modeling approaches, leading to further limitations and uncertainties. The energy scenarios used in the foreground (FG) and background (BG) approaches are from 2017 and could be outdated. The BG scenarios are far from a possible reality—due to the assumption that one energy mix represents the energy mix for all countries. However, they are useful to measure the impact of different energy mixes and to determine the best possible performance, e.g., by using the most environmentally beneficial energy provider available. The IAM scenarios depend on the REMIND model results, its modeling and data assumptions and the allocation in ecoinvent made by the *premise* tool. While the BG and FG scenarios refer exclusively to the energy sector, the IAM scenarios additionally modify the industry sector, which leads to an inconsistency in the comparison. On the contrary, it allows measuring the impact of the changes beyond the energy sector.

From a methodological perspective, there are further limitations: The cradle-to-gate approach of the LCA limits, the validity of the climate change impact assessment results due to the unknown fate of the CO_2 bound in the potential products. If the products are further processed into long-lasting products, the presented net negative climate change effects could apply. However, if they are further synthesized, e.g., into fuels, and combusted, the CO_2 is released and thus has no negative carbon footprint. Furthermore, since time was not included in the functional unit, the AP system has the disadvantage of being dependent on fluctuating energy from the sun, i.e., the amount of product per hour cannot be regulated on demand. Conventional systems have the disadvantage of depending on exhaustible fossil energy sources, which was also not included in the assessment. Further uncertainties, approximations, and assumptions arise from the database used, the database modifications, and the LCIA methods, which may have an influence on the results and the scenario comparison.

In addition, only two midpoint impact categories were chosen to limit the scope of the study, which is not a full environmental assessment. Therefore, the LCIA results for AP are a first estimate limited by the specific process pathway and its simplified modeling with a very low TRL and thus cannot be used for comprehensive comparisons and conclusions about its environmental performance. Nevertheless, the results allow us to give a first estimate of the climate change and land use impacts of AP and its sensitivity to background data modifications through different approaches and scenarios.

4.5 Conclusion and Outlook

Under the assumptions made and the limitations given, we conclude that the application of different scenario approaches in prospective LCA has a major impact on the LCIA results in the investigated impact categories climate change and land use. The impact assessment results of both impact categories show a similar trend with the scenario modifications in all product systems. For the investigated reaction pathways, the scenario variance had no major impact on the technology ranking of the AP and the reference product system. Nevertheless, the improvement in the climate change impact category of the new technology compared to the reference technology may be underestimated without the adjustment of the background database (up to 65% between FG and IAM). Hence, the environmental benefits of AP products in terms of climate change impacts are probably higher accompanied by a transformation in the energy and industry sectors, as modeled in the IAM scenarios. However, this needs to be confirmed by more detailed AP models as well as assessments of other emerging technologies.

The trade-off between climate change and land use impacts is reduced when applying IAM scenarios. In particular, the future 2050-IAM scenarios reduced land use impacts by up to 83% compared to foreground data modification alone. Nevertheless, land use impacts remain high in AP systems in all scenarios and thus further research is needed to investigate combined land use approaches with modified characterization factors for the land use impact assessment method.

The sensitivity analysis showed that the uncertainty of solar-to-chemical energy efficiency can reach up to 221%/79% lower climate change/land use impacts when varied from 1 to 5%. This shows that the parameter sensitivity in the case of AP, particularly to the solar-to-chemical energy efficiency and the data reliability of the photoreactor can have an equal or a more notable effect on the considered LCIA results than the background data modification.

From the LCIA variance, the performance difference between the AP and the reference product system, and the decrease in the trade-off between climate change and land use impacts we conclude, that scenario approaches with a background data modification have advantages over foreground modification alone. Although the BG scenario approach is advantageous for specific research questions (e.g., for the product footprint with a certain electricity mix), it can lead to extreme results, while the IAM scenario approach provides more moderate results and a more detailed description of the possible future background system. For emerging technologies, particularly AP, we therefore recommend IAM scenario approaches, but further research is needed here to test this approach with other technologies and to provide a prospective basis for comparison between emerging technologies.

Acknowledgements The authors gratefully acknowledge the fruitful discussion with Jennifer Strunk, Reinhard Schomäcker and Nikolaos G. Moustakas on the topic of photocatalysis.

References

Artz J, Müller TE, Thenert K, Kleinekorte J, Meys R, Sternberg A, Bardow A, Leitner W (2018) Sustainable conversion of carbon dioxide: an integrated review of catalysis and life cycle assessment. Chem Rev 118:434–504. https://doi.org/10.1021/acs.chemrev.7b00435

Barber J, Tran PD (2013) From natural to artificial photosynthesis. J R Soc Interface 10:20120984. https://doi.org/10.1098/rsif.2012.0984

Baumstark L, Bauer N, Benke F, Bertram C, Bi S, Gong CC, Dietrich JP, Dirnaichner A, Giannousakis A, Hilaire J, Klein D, Koch J, Leimbach M, Levesque A, Madeddu S, Malik A, Merfort A, Merfort L, Odenweller A, Pehl M, Pietzcker RC, Piontek F, Rauner S, Rodrigues R, Rottoli M, Schreyer F, Schultes A, Soergel B, Soergel D, Strefler J, Ueckerdt F, Kriegler E, Luderer G (2021) REMIND2.1: transformation and innovation dynamics of the energy-economic system within climate and sustainability limits. Geosci Model Dev 14:6571–6603. https://doi.org/10.5194/gmd-2021-85

Bergerson JA, Brandt A, Cresko J, Carbajales-Dale M, MacLean HL, Matthews HS, McCoy S, McManus M, Miller SA, Morrow WR, Posen ID, Seager T, Skone T, Sleep S (2020) Life cycle assessment of emerging technologies: evaluation techniques at different stages of market and technical maturity. J Ind Ecol 24:11–25. https://doi.org/10.1111/jiec.12954

Blankenship RE, Tiede DM, Barber J, Brudvig GW, Fleming G, Ghirardi M, Gunner MR, Junge W, Kramer DM, Melis A, Moore TA, Moser CC, Nocera DG, Nozik AJ, Ort DR, Parson WW, Prince RC, Sayre RT (2011) Comparing photosynthetic and photovoltaic efficiencies and recognizing the potential for improvement. Science 332:805–809. https://doi.org/10.1126/science.1200165

Bonke SA, Wiechen M, MacFarlane DR, Spiccia L (2015) Renewable fuels from concentrated solar power: towards practical artificial photosynthesis. Energy Environ Sci 8:2791–2796. https://doi.org/10.1039/C5EE02214B

Braham RJ, Harris AT (2009) Review of major design and scale-up considerations for solar photocatalytic reactors. Ind Eng Chem Res 48:8890–8905. https://doi.org/10.1021/ie900859z

Chen Z, Jaramillo TF, Deutsch TG, Kleiman-Shwarsctein A, Forman AJ, Gaillard N, Garland R, Takanabe K, Heske C, Sunkara M, McFarland EW, Domen K, Miller EL, Turner JA, Dinh HN (2010) Accelerating materials development for photoelectrochemical hydrogen production: standards for methods, definitions, and reporting protocols. J Mater Res 25:3–16. https://doi.org/10.1557/JMR.2010.0020

Cox B, Mutel CL, Bauer C, Mendoza Beltran A, van Vuuren DP (2018) Uncertain environmental footprint of current and future battery electric vehicles. Environ Sci Technol 52:4989–4995. https://doi.org/10.1021/acs.est.8b0026

Dincer I, Acar C (2015) Review and evaluation of hydrogen production methods for better sustainability. Int J Hydrog Energy 40:11094–11111. https://doi.org/10.1016/j.ijhydene.2014.12.035

Dufour J, Serrano DP, Gálvez JL, González A, Soria E, Fierro JL (2012) Life cycle assessment of alternatives for hydrogen production from renewable and fossil sources. Int J Hydrog Energy 37:1173–1183. https://doi.org/10.1016/j.ijhydene.2011.09.135

European Commission (2010) Joint Research Centre. Institute for Environment and Sustainability. International Reference Life Cycle Data System (ILCD). Handbook—general guide for Life cycle assessment—detailed guidance (1st edn). EUR 24708. Publications Office of the European Union, Luxembourg

European Commission (2011) Joint Research Centre. Institute for Environment and Sustainability. Recommendations for life cycle impact assessment in the European context: Institute for Environment and Sustainability: International Reference Life Cycle Data System (ILCD). Handbook—recommendations for life cycle impact assessment in the European context. EUR 24571 EN. Publications Office of the European Union, Luxembourg

European Commission (2021) Photovoltaic geographical information system. https://re.jrc.ec.europa.eu/pvg_tools/en/#MR. Accessed 24 Aug 2021

Fujishima A, Honda K (1972) Electrochemical photolysis of water at a semiconductor electrode. Nature 238:37–38. https://doi.org/10.1038/238037a0
Fuss M, Xu L (2021) Unintended environmental impacts at local and global scale—trade-offs of a low-carbon electricity system. In: Möst D, Schreiber S, Herbst A, Jakob M, Martino A, Poganietz W-R (eds) The future European energy system: renewable energy, flexibility options and technological progress, 1st edn. Springer International Publishing; Imprint Springer, Cham, pp 237–255
Hoffmann MR, Moss JA, Baum MM (2011) Artificial photosynthesis: semiconductor photocatalytic fixation of CO2 to afford higher organic compounds. Dalton Trans 40:5151–5158. https://doi.org/10.1039/C0DT01777A
International Standard Organization (2006a) ISO 14040:2006a, ISO/TC 207/SC 5. Environmental management—life cycle assessment—principles and framework. International Organization for Standardization, Geneva
International Standard Organization (2006b) ISO 14044:2006b, ISO/TC 207/SC 5. Environmental management—life cycle assessment—requirements and guidelines. International Organization for Standardization, Geneva
IPCC (2019) Global Warming of 1.5°C. In: Masson-Delmotte V, Zhai P, Pörtner H-O, Roberts D, Skea J, Shukla PR, Pirani A, Moufouma-Okia W, Péan C, Pidcock R, Connors S, Matthews JBR, Chen Y, Zhou X, Gomis MI, Lonnoy E, Maycock T, Tignor M, Waterfield T (eds) An IPCC special report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. In Press
IPCC (2021) Climate change 2021: the physical science basis. In: Masson-Delmotte V, Zhai P, Pirani A, Connors SL, Péan C, Berger S, Caud N, Chen Y, Goldfarb L, Gomis MI, Huang M, Leitzell K, Lonnoy E, Matthews JBR, Maycock TK, Waterfield T, Yelekçi O, Yu R, Zhou B (eds) Contribution of working group I to the sixth assessment report of the intergovernmental panel on climate change. Cambridge University Press
Ioannou I, D'Angelo SCC, Galán-Martín Á, Pozo C, Perez-Ramirez J, Guillén-Gosálbez G (2021) Process modelling and life cycle assessment coupled with experimental work to shape the future sustainable production of chemicals and fuels. React Chem Eng https://doi.org/10.1039/D0RE00451K
Joyce PJ, Björklund A (2021) Futura: a new tool for transparent and shareable scenario analysis in prospective life cycle assessment. J Ind Ecol. https://doi.org/10.1111/jiec.13115
Karamian E, Sharifnia S (2016) On the general mechanism of photocatalytic reduction of CO2. J CO2 Util 16:194–203. https://doi.org/10.1016/j.jcou.2016.07.004
Kim J, Jeong S, Beak M, Park J, Kwon K (2022) Performance of photovoltaic-driven electrochemical cell systems for CO2 reduction. Chem Eng J 428:130259
Larkum AWD (2010) Limitations and prospects of natural photosynthesis for bioenergy production. Curr Opin Biotechnol 21:271–276. https://doi.org/10.1016/j.copbio.2010.03.004
de Lasa HI, Serrano B, Salaices M (2005) Photocatalytic reaction engineering. Springer, New York
Lassner E, Schubert W-D (1999) Tungsten: properties, chemistry, technology of the element, alloys, and chemical compounds. Springer, New York
Laurent A, Espinosa N, Hauschild MZ (2018) LCA of energy systems. In: Hauschild MZ, Rosenbaum RK, Olsen SI (eds) Life cycle assessment: theory and practice. Springer, Cham, pp 633–668
Luderer G, Leimbach M, Bauer N, Kriegler E, Aboumahboub T, Curras TA, Baumstark L, Bertram C, Giannousakis A, Hilaire J, Klein D, Mouratiadou I, Pietzcker R, Piontek F, Roming N, Schultes A, Schwanitz VJ, Strefler J (2013) Description of the REMIND Model (Version 1.5). SSRN J. https://doi.org/10.2139/ssrn.2312844
Maes B, Audenaert A, Craeye B, Buyle M (2021) Consequential exante life cycle assessment on clinker production in the EU: how will the future influence its environmental impact? J Cleaner Prod 315:128081. https://doi.org/10.1016/j.jclepro.2021.128081

Marini C, Blanc I (2014) Towards prospective life cycle assessment: how to identify key parameters inducing most uncertainties in the future? Application to photovoltaic systems installed in Spain. In: Hutchison D, Kanade T, Kittler J, Kleinberg JM, Kobsa A, Mattern F, Mitchell JC, Naor M, Nierstrasz O, Pandu Rangan C, Steffen B, Terzopoulos D, Tygar D, Weikum G, Murgante B, Misra S, Rocha AMAC, Torre C, Rocha JG, Falcão MI, Taniar D, Apduhan BO, Gervasi O (eds) Computational science and its applications – ICCSA 2014: 14th international conference, Guimarães, Portugal, June 30–July 3, 2014; proceedings, vol 8581. Springer, Cham, pp 691–706

May MM, Rehfeld K (2019) ESD ideas: photoelectrochemical carbon removal as negative emission technology. Earth Syst Dyn 10:1–7. https://doi.org/10.5194/esd-10-1-2019

Mendoza BA, Cox B, Mutel C, Vuuren DP, Font Vivanco D, Deetman S, Edelenbosch OY, Guinée J, Tukker A (2020) When the background matters: using scenarios from integrated assessment models in prospective life cycle assessment. J Ind Ecol 24:64–79. https://doi.org/10.1111/jiec.12825

Mutel C (2017) Brightway: an open source framework for life cycle assessment. JOSS 2:236. https://doi.org/10.21105/joss.00236

Mutel C, Cox B (2021) Wurst. https://github.com/polca/wurst. Accessed 3 Oct 2021

National Institute of Standards and Technology (2020) Computational chemistry comparison and benchmark database: standard reference database 101. https://cccbdb.nist.gov/exp1x.asp. Accessed 6 Oct 2021

National Institute of Standards and Technology (2021) NIST chemistry webbook: NIST standard reference database number 69. https://webbook.nist.gov/chemistry/. Accessed 24 Aug 2021

Nuss P, Eckelman MJ (2014) Life cycle assessment of metals: a scientific synthesis. PLoS One 9:e101298. https://doi.org/10.1371/journal.pone.0101298

Parvatker AG, Eckelman MJ (2019) Comparative evaluation of chemical life cycle inventory generation methods and implications for life cycle assessment results. ACS Sustain Chem Eng 7:350–367. https://doi.org/10.1021/acssuschemeng.8b03656

Patyk A, Poncette D, Lazar L, Wendeberg E (2020) PROPHECY - Prozesskonzepte für die Photokatalytische CO2-Reduktion verbunden mit Life-Cycle-Analysis: Schlussbericht zum Teilvorhaben C Effizienz- und Nachhaltigkeitsoptimierung, Karlsruhe, Germany. Unpublished

Parisi ML, Maranghi S, Vesce L, Sinicropi A, Di Carlo A, Basosi R (2020) Prospective life cycle assessment of third-generation photovoltaics at the pre-industrial scale: a long-term scenario approach. Renew Sustain Energy Rev 121:109703. https://doi.org/10.1016/j.rser.2020.109703

Paul Scherrer Institut, Potsdam Institute for Climate Impact Research (2022) Premise: User guide. https://premise.readthedocs.io/en/latest/. Accessed 3 May 2022

Pizzol M, Sacchi R, Köhler S, Anderson Erjavec A (2021) Non-linearity in the life cycle assessment of scalable and emerging technologies. Front Sustain 1 https://doi.org/10.3389/frsus.2020.611593

Pfluger B, Testeegen B, Franke B (2017) Langfristszenarien für die Transformation des Energiesystems in Deutschland: Modul 3: Referenzszenario und Basisszenario Studie im Auftrag des Bundesministeriums für Wirtschaft und Energie. https://www.bmwi.de/Redaktion/DE/Downloads/B/berichtsmodul-3-referenzszenario-und-basisszenario.pdf%3F__blob%3DpublicationFile%26v%3D4. Accessed 7 July 2022

RWTH Aachen (2014) Schlussbericht CO2RRECT - Verwertung von CO2 als Kohlenstoffbaustein unter Verwendung überwiegend renerativer Energie. https://www.tib.eu/de/suchen/id/TIBKAT%3A792621247/CO2RRECT-Schlussbericht-Berichtszeitraum-01-10/. Accessed 7 July 2022

Sacchi R (2020) Coupling integrated assessment models and ecoinvent for prospective environmental impact assessment. https://premise.readthedocs.io/en/latest/introduction.html. Accessed 1 Sep 2021

Sacchi R, Terlouw T, Siala K, Dirnaichner A, Bauer C, Cox B, Mutel C, Daioglou V, Luderer G (2022) PRospective EnvironMental Impact asSEment (premise): a streamlined approach to producing databases for prospective life cycle assessment using integrated assessment models. Renew Sustain Energy Rev 112311. https://doi.org/10.1016/j.rser.2022.112311

Sathre R, Scown CD, Morrow WR, Stevens JC, Sharp ID, Ager JW, Walczak K, Houle FA, Greenblatt JB (2014) Life-cycle net energy assessment of large-scale hydrogen production via photoelectrochemical water splitting. Energy Environ Sci 7:3264–3278. https://doi.org/10.1039/C4EE01019A
Sathre R, Greenblatt JB, Walczak K, Sharp ID, Stevens JC, Ager JW, Houle FA (2016) Opportunities to improve the net energy performance of photoelectrochemical water-splitting technology. Energy Environ Sci 9:803–819. https://doi.org/10.1039/C5EE03040D
Schakel W, Oreggioni G, Singh B, Strømman A, Ramírez A (2016) Assessing the techno-environmental performance of CO2 utilization via dry reforming of methane for the production of dimethyl ether. J CO2 Util 16:138–149. https://doi.org/10.1016/j.jcou.2016.06.005
Schreiber A, Zapp P, Kuckshinrichs W (2009) Environmental assessment of German electricity generation from coal-fired power plants with amine-based carbon capture. Int J Life Cycle Assess 14:547–559. https://doi.org/10.1007/s11367-009-0102-8
Schreiber A, Zapp P, Markewitz P, Vögele S (2010) Environmental analysis of a German strategy for carbon capture and storage of coal power plants. Energy Policy 38:7873–7883. https://doi.org/10.1016/j.enpol.2010.09.006
Seo H, Katcher MH, Jamison TF (2017) Photoredox activation of carbon dioxide for amino acid synthesis in continuous flow. Nat Chem 9:453–456. https://doi.org/10.1038/nchem.2690
Sternberg A, Bardow A (2015) Power-to-what?—environmental assessment of energy storage systems. Energy Environ Sci 8:389–400. https://doi.org/10.1039/C4EE03051F
Sternberg A, Bardow A (2016) Life cycle assessment of power-to-gas: syngas vs methane. ACS Sustain Chem Eng 4:4156–4165. https://doi.org/10.1021/acssuschemeng.6b00644
Steubing B, de Koning D, Haas A, Mutel CL (2020) The activity browser—an open source LCA software building on top of the brightway framework. Softw Impacts 3:100012. https://doi.org/10.1016/j.simpa.2019.100012
Tahir M, Amin NS (2013) Recycling of carbon dioxide to renewable fuels by photocatalysis: prospects and challenges. Renew Sustain Energy Rev 25:560–579. https://doi.org/10.1016/j.rser.2013.05.027
Telsnig T (2015) Standortabhängige Analyse und Bewertung solarthermischer Kraftwerke am Beispiel Südafrikas. Dissertation, University of Stuttgart
Terlouw T, Bauer C, Rosa L, Mazzotti M (2021a) Life cycle assessment of carbon dioxide removal technologies: a critical review. Energy Environ Sci 14:1701–1721. https://doi.org/10.1039/D0EE03757E
Terlouw T, Treyer K, Bauer C, Mazzotti M (2021b) Life cycle assessment of direct air carbon capture and storage with low-carbon energy sources. Environ Sci Technol 55:11397–11411. https://doi.org/10.1021/acs.est.1c03263
Thonemann N, Schulte A, Maga D (2020) How to conduct prospective life cycle assessment for emerging technologies? A systematic review and methodological guidance. Sustainability 12:1192. https://doi.org/10.3390/su12031192
Trudewind CA, Schreiber A, Haumann D (2014) Photocatalytic methanol and methane production using captured CO2 from coal-fired power plants. Part I—a Life Cycle Assessment. J Cleaner Prod 70:27–37. https://doi.org/10.1016/j.jclepro.2014.02.014
Vahidi E, Zhao F (2017) Environmental life cycle assessment on the separation of rare earth oxides through solvent extraction. J Environ Manag 203:255–263. https://doi.org/10.1016/j.jenvman.2017.07.076
Vidal O, Goffé B, Arndt N (2013) Metals for a low-carbon society. Nat Geosci 6:894–896. https://doi.org/10.1038/ngeo1993
Volkart K, Bauer C, Boulet C (2013) Life cycle assessment of carbon capture and storage in power generation and industry in Europe. Int J Greenhouse Gas Control 16:91–106. https://doi.org/10.1016/j.ijggc.2013.03.003
Weidema BP, Bauer C, Hischier R, Mutel C, Nemecek T, Reinhard J, Vadenbo CO, Wernet G (2013) Overview and methodology: data quality guideline for the ecoinvent database version 3. Ecoinvent Report 1 (v3). The Ecoinvent Centre, St. Gallen

Wenham S (2009) Applied photovoltaics, 2nd edn. Earthscan, London
Wernet G, Bauer C, Steubing B, Reinhard J, Moreno-Ruiz E, Weidema B (2016) The ecoinvent database version 3 (part I): overview and methodology. Int J Life Cycle Assess 21:1218–1230. https://doi.org/10.1007/s11367-016-1087-8
Yang W, Zhang J-H, Si R, Cao L-M, Zhong D-C, Lu T-B (2021) Efficient and steady production of 1: 2 syngas (CO/H2) by simultaneous electrochemical reduction of CO2 and H2O. Inorg Chem Front 8:1695–1701. https://doi.org/10.1039/D0QI01351J
Zhai P, Haussener S, Ager J, Sathre R, Walczak K, Greenblatt J, McKone T (2013) Net primary energy balance of a solar-driven photoelectrochemical water-splitting device. Energy Environ Sci 6:2380. https://doi.org/10.1039/C3EE40880A

Chapter 5
Understanding Soil Organic Carbon Dynamics of Short Rotation Plantations After Land Use Change—From Establishment to Recultivation

Enrique Alejandro Perdomo Echenique and Franziska Hesser

Abstract The increase in soil organic carbon (SOC) stocks has the potential to contribute to climate mitigation strategies by reducing atmospheric CO_2. Short rotation plantations (SRP) provide bio-based resources and can possibly accumulate SOC. Estimating the potential SOC stocks of short rotation plantations can help decision-makers to implement strategies that reduce SOC loss and thus contribute to climate change mitigation. The dynamic changes in SOC were estimated for a case study using the RothC carbon turnover model. The results indicate that SOC stocks increased from 37.8 to 48.52 t C/ha within 20 years of the plantation's lifetime. Thus, an annual average increase of 0.535 t C/ha year is expected. Given the importance of implementing strategies that support the potential climate mitigation benefits of SRP, a sensitivity analysis was employed to identify the relevant factors that affected SOC prediction. For instance, the influence of soil condition heterogeneity, such as clay content, can vary the estimations of SOC accumulated. This highlights the relevance of obtaining primary data at different locations within the plantation's areas: to obtain a variety of SOC stock estimations that give a better representation of SOC accumulation. Such analysis help to propose suggestions that mitigate the climate effect of short rotation plantations.

Keywords Life cycle assessment · Soil organic carbon · Short rotation plantation · Poplar · Land use change · Wood · Bioeconomy

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/978-3-031-29294-1_5.

E. A. Perdomo Echenique (✉)
Institute of Marketing and Innovation, University of Natural Resources and Life Sciences, Vienna, Feistmantelstraße 4, 1180 Vienna, Austria
e-mail: alejandro.perdomo@boku.ac.at

F. Hesser
Wood K plus – Competence Center for Wood Composites and Wood Chemistry, Kompetenzzentrum Holz GmbH, Altenberger Straße 69, 4040 Linz, Austria

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*,
Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_5

5.1 Introduction

In light of the global need to deal with the predicted consequences of anthropogenic generated climate change, it has become imperative to implement strategies for atmospheric CO_2 reduction (IPCC 2022). An increase in global soil organic carbon (SOC) stocks is considered one of the most promising and important climate change mitigation (CCM) strategies to date (Minasny et al. 2017), as indicated at the 21st Conference of the Parties in Paris in 2015, where the "4 per 1000 soils for Food Security and Climate" initiative was launched. It was proposed that with an annual increase of 0.4% of SOC stocks in the top soil layers (within 1 m), a reduction of 20–35% of global anthropogenic greenhouse gasses (GHG) emissions could be achieved, allowing nations committed to the Paris accord to make significant strides towards that goal (Fantin et al. 2022; Rumpel et al. 2020).

Considering the increase in biomass demand for material use and renewable energy (Schmidt-Walter 2019), SRP might help to provide another source of woody material (Buchholz et al. 2005; Zanchi et al. 2013). The cultivation of fast-growing trees, such as poplar (*Populus*) in SRP like short rotation coppice, agricultural wood production, short rotation wood cultivation) has been presented as an attractive agricultural practice to provide woody bio-mass and simultaneously increment SOC stocks, particularly when substituting previous agricultural land or marginal land (Don et al. 2012). Compared to annual crops, SRP are perennial crops that lower soil disturbance, which allows better incorporation of root and leaf litter into the soil, consequently maintaining or generating SOC accumulation (Don et al. 2012).

Growing number of scientific research studies on Life Cycle Assessments (LCA) of SRPs and their potential contribution to climate change mitigation demonstrate the relevance of estimating SOC dynamics to evaluate the mitigation potential and develop strategies that support the sustainable management of SRP (Clarke et al. 2019; Lockwell et al. 2012; Petersen et al. 2013; Rytter et al. 2015). For instance, by identifying the agricultural practices that affect SOC accumulation, strategies to increase SOC in SRP can be deduced. A main issue in deducing such strategies is the challenge of calculating SOC ex-ante to SRP establishment when primary data is still absent (Rowe et al. 2020). As for previous LCA studies (e.g., Barancikova et al. 2010; Berhongaray et al. 2017), the focus of this study is mainly on SOC computation, however, with the specific goal of closing the knowledge gap regarding modeling of SOC for SRP ex-ante to its establishment.

Methods for assessing SOC changes in agricultural LCA have been previously classified between those based on observation, emissions factors, and simulation models (Goglio et al. 2015. Observation methods are grounded in direct field measurements, providing the highest certainty level of primary data for LCAs (e.g., McClean 2014; Pacaldo et al. 2013). However, the long-term nature of SRP and the inherent required time (e.g., one year) to accumulate SOC, inhibit performing direct measurements and consequently they are not viable for early assessments. Emissions factors, such as those used by the IPCC Tier 1 method (IPCC 2019), were developed from national and international statistical data; however, they lack spatial and temporal precision to account for site-specific soil and climate characteristics (Hillier et al. 2009). Alternatively, carbon turnover (CT) simulation models (e.g., RothC) allow for the integration of soil (e.g., clay content) and climate (e.g., temperature changes) effects. Goglio et al. (2015) classified CT models between Simple Carbon Models (SCM), and Dynamic Crop-climate-soil Models (DCM). SCM involves a simpler set of equations, as they do not include crop production interactions and are usually operated on a monthly or yearly basis. DCM includes the interactions between crop production, SOC change, Nitrogen cycles, and GHG emissions on a daily time-step basis (e.g., CENTURY, DNDC). Nevertheless, for both SCM and DCM, the results are fully dependent on the model's calibration and the data entry into the models (Goglio et al. 2015; Rampazzo Todorovic et al. 2010). In comparison to DCM, SCM are easier to implement due to lower data requirement. Such A characteristic makes SCM an attractive option for estimating soil carbon accumulation ex-ante to SRP establishment or in early phases. Nevertheless, there are at least three critical challenges that make implementing models to quantify soil organic carbon dynamics/stocks difficult in the context of recently established SRP and SOC predictions: (1) SCT model accuracy depends on the availability and quality of data to calibrate the models to similar field conditions (Ericsson 2015; Fantin et al. 2022); (2) Heterogeneity of local conditions (e.g., soil type, microclimate) and its spatial effects (Goglio et al. 2015); (3) Accounting for the plantation's lifetime (Harris et al. 2015).

The present study aimed to analyze these three challenges, as well as deliver suggestions on how to deal with them in the context of implementing SCT models for predicting SOC dynamics, and subsequent effects on climate impacts of Poplar SRP during the early stages of establishment. Moreover, by predicting the SOC dynamics at an early stage, we attempted to deduce agricultural strategies that support SOC accumulation in SRP. Therefore, a case study of a Slovakian SRP was studied. The case investigated is part of the European funded demonstration project Dendromass for Europe (D4EU), which aims to establish sustainable SRP-based regional dendromass cropping systems on marginal land that feeds its dendromass into the production of bio-based products.

The specific objectives of the study are to:

1. Calculate the potential SOC accumulation of Poplar under the site conditions at Brodské, Slovakia by using a SCT model.
2. Assess the climate impacts from growing Poplar SRC in a 5, 10 and 20-year timespan.
3. Identify the factors that affect predicting SOC dynamics during the early stages of an SRC value chain.

5.2 Methods

The objectives of the study were addressed through the combination of a literature review and experiences from a case study described below (Sects. 5.2.1 and 5.2.2). The carbon turnover model RothC V.26.3 by Coleman and Jenkinson (2014) was used to model SOC levels. The RothC model was selected since it is one of the most used SCM implemented in LCA (Albers et al. 2020). Subsequently, the results were combined with the results of an LCA study (Perdomo et al. 2021) to estimate the climate impacts of growing Poplar SRC in a 5-, 10- and 20-year timespan, which is the five-year rotation cycle for harvesting the woody material. The climate impacts were assessed based on the Global Warming Potential 100 (GWP_{100}) indicator. Within the following sub-sections, the case study and the SOC calculations are described.

5.2.1 System Description

The Poplar SRP was planned for a cultivation of 20 years with a five-year harvesting cycle. As illustrated in Fig. 5.1, the first operational step is preparing the land for planting the poplar rods with the activities of heavy disking, ploughing, and harrowing. The rods were planted by combining manual labor and machinery. During the first four years of plantation, disk harrowing was necessary for weed control. Singling and partially pruning were done manually to select the supporting dominant shoot, as such a step was necessary after every harvest. In the harvesting phase, a multi-stem fully mechanized harvesting approach was implemented every fifth year. The last harvest is done after 20 years to complete four harvesting cycles. It is assumed that the end of the life of the plantation is constituted by ploughing the land and extracting the wood stems and roots so that the land can be converted back to annual arable cultivation conditions. It must be noted that the effects of recultivating the land after 20 years of SRP cultivation were only accounted for in the calculations of the potential environmental impacts of the agricultural operations. The consequences of ploughing the land, wood stems, and root extraction were not considered within the SOC stock estimation. Such exclusion presents a limitation of the present study, and it is justified by the lack of SOC primary data. Firstly, the collection of such data was not possible since the plantation's end of life has not been reached yet, making the data collection unviable. Secondly, as this data is not available, it was decided to not model the end of life effects as the SCM RothC V.26.3 carbon turnover model needs reference data to reduce the uncertainties of the results. Nevertheless, the relevance of the impacts of plantations' end of life on SOC is deliberated within the discussion section.

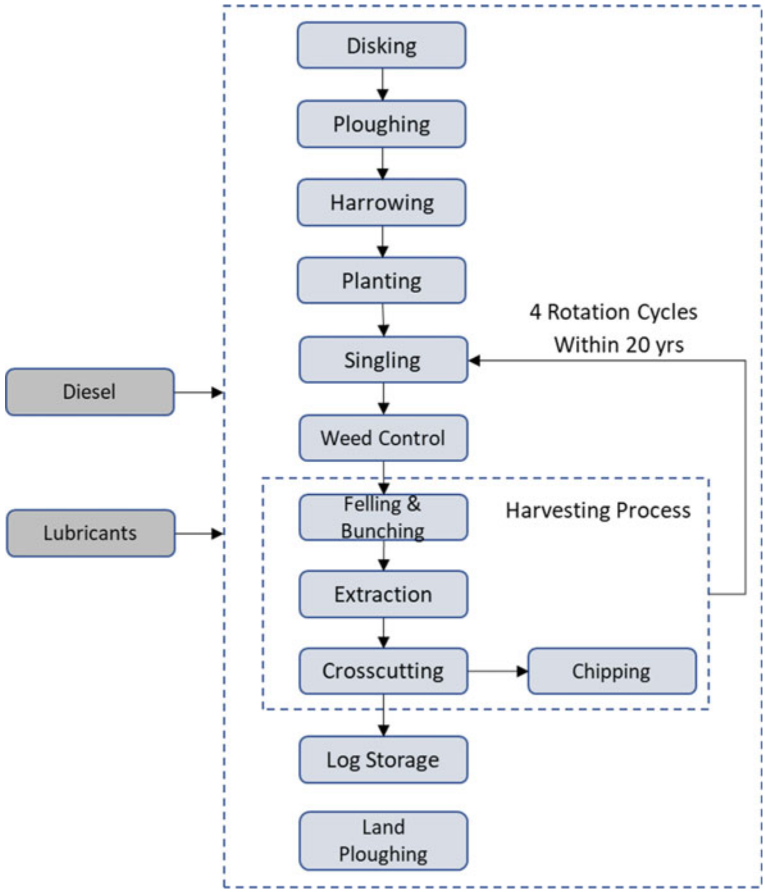


Fig. 5.1 Flow diagram for SRP production system

5.2.2 Site Conditions and Field Data

The SRC plantation is in Malacky, within the Pannonian Basin. Primary climate data, such as temperature and precipitation (Table 5.1), were measured in Brodské, Slovakia (Fontenla-Razzetto et al. 2022). Since Brodské and Malacky are located within the Pannonian Basin it was assumed that the climate data were transferable for both areas, since both areas are within the same biogeographic region. The former land use was a cornfield, which presented an SOC content at 0–30 cm soil depth of 37.8 t ha^{-1}, with a bulk soil density of 1.25 t (m^3)$^{-1}$, and 4.9% clay content (Rossi 2018). Field trails estimated an average poplar yield of 8.1 dry t ha^{-1}.

Table 5.1 Climate data used for RothC V.26.3 (modified from Fürtner et al. 2022)

Date	Average mean temperature (°C)	Average mean precipitation (mm)	Average mean ETP Penman–Monteith (mm)
May.18	20.40	1.00	4.107
Jul.18	20.95	1.23	4,002
Sep.18	16.03	3.56	2.02
Nov.18	9.02	0.47	0.34
Jan.19	0.17	1.51	0.43
Mar.19	7.17	0.76	0.97
Jun.19	22.02	1.06	4.56
Aug.19	21.17	1.69	3.25
Oct.19	10.86	0.92	0.68
Dec.19	2.52	1.65	0.16
Feb.20	5.59	1.0	0.84
Mar.20	5.85	0.76	1.39

5.2.3 Soil Organic Carbon Modeling

The RothC V.26.3 carbon turnover model was used to estimate the SOC levels during a 20-year timespan. The model has been used to evaluate carbon turnover in arable soils in England; nevertheless, its use has extended to other ecosystems by calibrating the model to site-dependent conditions (Barancikova et al. 2010). SOC in non-waterlogged surface soils is calculated in a monthly time step as a function of vegetation cover, climate conditions, soil type, and soil management. The model is based on the physical–chemical interactions of four active pools, such as Resistant Plant Material (RPM); Decomposable Plant Material (DPM); Microbial Biomass (BIO); Humified Organic Matter (HUM), and one inactive pool Inert Organic Matter (IOM), which is not involved in the turnover processes. The RothC model was calibrated to the SRP site climate condition based on the procedures Rampazzo Todorovic et al. (2010) described. Thus, the RothC model was translated to a Microsoft Excel spreadsheet, where inverse modeling was used to integrate the site climate conditions and calculate the model initialization parameters (RPM, DPM, RPM, and BIO) which are based on the distribution of initial SOC (SOC_{in}) (Morais et al. 2018). As presented in Fig. 5.2, before the calibration procedure, the measured SOC_{in} (SOC_{in} MS), % clay content, and climate data as Temperature (T), Pressure (P) and Evapotranspiration (ET) were entered into the model as constant variables. Afterward, the calibration was done by modifying the plant input data until the SOC_{in} (MS) and calculated SOC_{in} (SOC_{in} (C)) had reached a root square mean error below 0.5. After the match between SOC_{in} (MS) and SOC_{in} (C) was achieved, the new model initialization parameters (IOM, RPM, HUM and BIO) were utilized to represent the distribution of organic C in the soil pools. Figure 5.3 presents the calibration results.

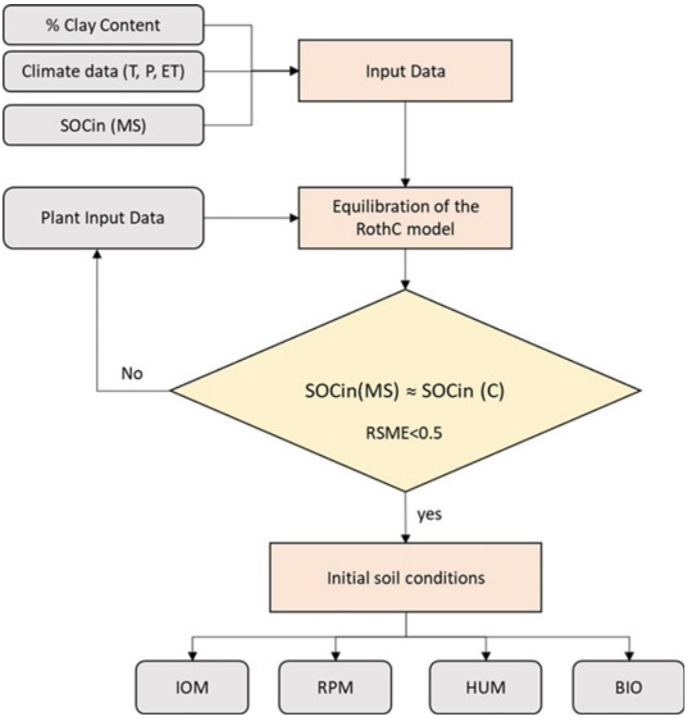


Fig. 5.2 Calibration process of RothC model

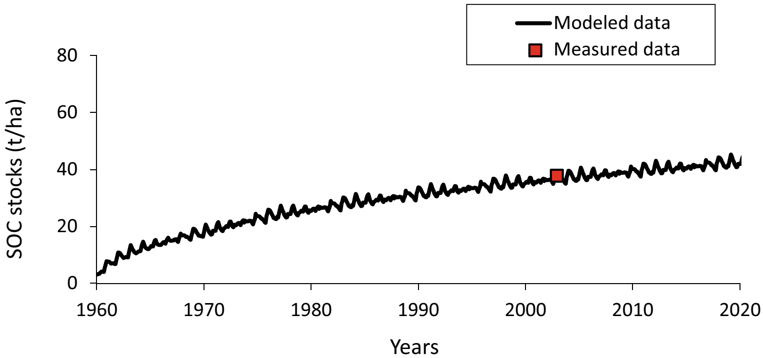


Fig. 5.3 Results of calibration of RothC model

SOC predictions were calculated after the carbon turnover model was calibrated. The main input data were plant carbon input, which was divided between belowground and aboveground carbon input (BGC and AGC). For the AGC, it was assumed that during the harvesting event, no woody material was left on the field; thus, the only carbon input comes from the leaf material. As no direct measurements for either AGC and BGC were available, it was necessary to estimate the inputs based on empirical equations and previous field data. Thus, following Eqs. 5.1, 5.2, and 5.3 by Gorgan and Matthews (2002) both carbon pool inputs were estimated. The procedure is described below.

$$\text{Aboveground input}\left(W_{Cin};\ t_C ha^{-1}\right):\quad W_{Cin} = \frac{LAI \cdot f_C}{SLA} + W_{AG}, f_{wa} \tag{5.1}$$

where,

W_{Cin} = Aboveground carbon input in tons (t_C) per ha (from leaves and wood);

LAI = Leaf Area Index;

f_C = fraction of carbon in leaves;

SLA = Specific Leaf Area;

W_{AG} = Carbon input from woody material (assumed to be zero);

f_{wa} = fraction of carbon in woody material.

The SLA and LAI were estimated by combining primary data from a previous field study (Heilig et al. 2020), and by corresponding regression analysis between variables. The field study provided the following: diameter at breast height (DBH), height (m), and leaf area index (LAI), as in Table 5.2 such values were used to estimate the missing LAI for the subsequent years (displayed as "?" in Table 5.2). The procedure is visualized in Figs. 5.4 and 5.5.

Parting from a similar procedure, Eq. 5.2 presents the method from Gorgan and Matthews (2002) to estimate BGC input. The first part of the equation represents input due to fine root turnover (root senescence, root respiration, and root rhizodeposition), whereas the second part considers the input due to death and decay of structural roots. Table 5.3 presents a summary of the data used to calculate the BGC input. As shown, a combination of literature and primary data was used. Primary data were available

Table 5.2 Relationship between measured and calculated tree parameters (Heilig et al. 2020)

Age	DBH (cm)	h (m)	Leaf area index
2.5	2.7	4.1	0.35
2.5	3	5.4	0.56
3.5	5.4	7	1.38
3.5	8.4	7.7	2.77
3.5	11.4	9.5	5.01
4	9.7	9.9	?
4	8.3	8.7	?
5	15.2	12	?
6	?	?	?

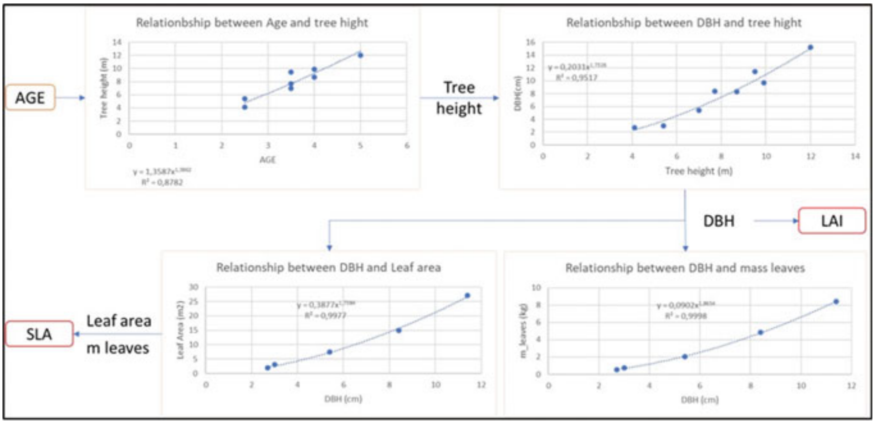


Fig. 5.4 Procedure for estimating SLA and LAI

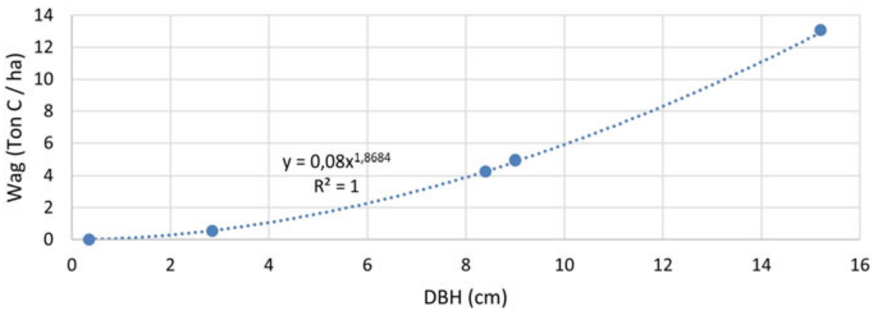


Fig. 5.5 Estimated Relationship between DBH and aboveground carbon input

from previous field studies.

$$\text{Belowground input } \left(W_{Rin};\ t_C ha^{-1}\right): \quad W_{Rin} = W_{yield},\ Fr,\ F_{frto} + W_{BG} \cdot f_{wb} \quad (5.2)$$

where, belowground carbon input in tons (t_C) per ha

W_{yield} = Above ground yield;

Table 5.3 Belowground carbon input data

F_{frto}	0.85	(Grogan P.* and Matthews 2002)
Fr	0.205	Primary data (Meyer M. et al., 2021)
W_{yield} (t_C ha^{-1} a^{-1})	3.85	Primary data provided by project partners
W_{BG} (t_C ha^{-1} a^{-1})	0.788	Primary data provided by project partners
T	5	Number of years since the last coppicing cycle

Fr = Root to Shoot Ratio;

F_{frto} = Fraction of below ground carbon lost due to fine root turnover;

W_{BG} = Weight of carbon below ground in the root system;

f_{wb} = fraction of the below ground carbon input that enters the fresh carbon.

The total plant carbon input C_i was estimated by summing the above and belowground carbon input as presented in Eq. 5.3.

$$\text{Total plant carbon input}\ \left(C_i;\ t_C ha^{-1}\right):\quad C_i = W_{Cin} + W_{Rin} \tag{5.3}$$

5.2.4 Sensitivity Analysis

Based on information from previous field studies and literature data, a sensitivity analysis was conducted by varying the value of one parameter, while maintaining the others constant. The main three observed parameters were soil conditions with the indicator of % clay, aboveground wood production, and time horizons. The parameters were selected in order to understand the effects of data variability, heterogeneity in local conditions, and different time horizons. Consequently, it was assumed that climate conditions would remain constant. Furthermore, the results of the sensitivity analysis scenarios were compared to the total climate impact from agricultural operations, which were previously estimated through an LCA study (Perdomo et al. 2021).

5.3 Results

5.3.1 Prediction of SOC

Starting from the initial SOC content of 37.8 t_C ha^{-1} before the land use changes from annual cropping to SRP, the results indicate that after 20 years there is SOC accumulation to about 48.52 t_C ha^{-1} (Fig. 5.6), corresponding to a total annual average increase of 0.535 t_C ha^{-1} a^{-1}. During the first four years of plantation, a decrease of approximately 9.1% in SOC occurred due to less carbon input from AGC and BGC compared to the carbon turnover processes of the SRC plantation. After the 5th year, SOC increased above the initial amount; however, during the first years after the first harvesting event, SOC stocks decreased. Approximately, after year 10, when the second harvest occurred, SOC stocks remained above the initial SOC levels. These results indicate that for the poplar SRC plantation case study, it takes approximately 10 years for the SOC stocks to be constantly above the initial SOC.

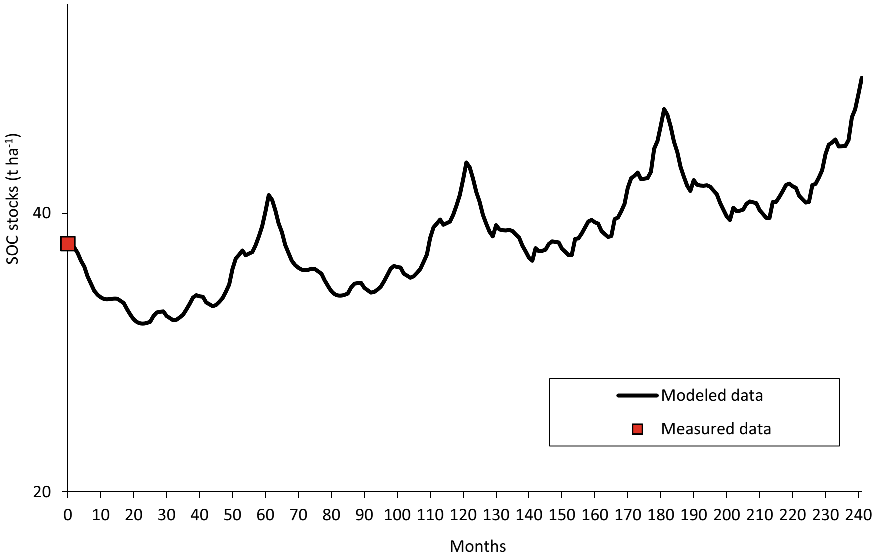
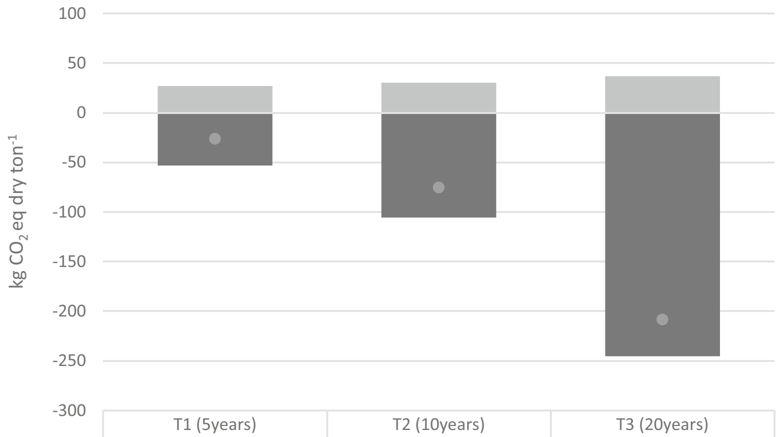


Fig. 5.6 SOC stocks during 20 years of plantation

5.3.2 Influence of Plantation Lifetime

The sensitivity analysis results for the net SOC stocks, the climate impacts from the agricultural operation, and the net carbon balance are shown in Fig. 5.7. A negative value of carbon balance means that there is a net decrease of GHG concentration in the atmosphere, consequently generating a CM effect. Considering the scenarios T1 (5 years), T2 (10 years), and T3 (20 years), the results (Fig. 5.7) show how the scenarios with the lowest lifetime have a lower SOC accumulation and consequently a smaller amount of net carbon mitigation. The influence of the plantations lifetime can be further understood by the results of the carbon fluxes (Fig. 5.8). During the first two years, and after each harvesting event, the carbon turnover processes generate higher amounts of emitted carbon to the atmosphere than the carbon sequestered by the trees. Thus, positive values (Fig. 5.8) of carbon flow (t ha^{-1}) occur. After the third year of the initial plantation and each harvesting event, the amount of carbon sequestered is higher than the emitted carbon. Moreover, it is between years four and five, when the negative carbon fluxes (Fig. 5.8) generate a higher SOC stock than the initial SOC (Fig. 5.6). We noticed that the increases in SOC accumulation through the plantation's lifetime outstand the emissions from the agricultural operation. Hence, the climate mitigation potential increases substantially together with the SRP lifetime since higher levels of SOC accumulation are achieved. It must be noted that the effects related to recultivation were only accounted for in the calculations of the potential environmental impacts of the agricultural operations—the SOC could not be modeled due to a lack of data.



	T1 (5years)	T2 (10years)	T3 (20years)
Agricultural operations	27	30	37
SOC	-53	-106	-245
Net Balance	-26	-75	-209

Fig. 5.7 Sensitivity analysis for plantation lifetime. Net carbon balance

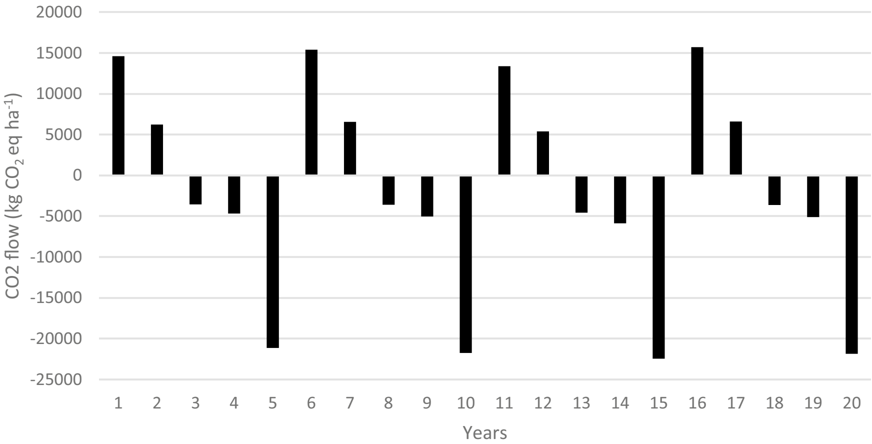


Fig. 5.8 Result of the carbon fluxes (difference between carbon emitted to the atmosphere and carbon sequestered)

5.3.3 *Influence of Clay Content*

The heterogeneity of soil conditions was assessed by changing the clay content between 3.7 and 10.60%. The scenarios analyzed are presented in Table 5.4. As a consequence of different clay contents, the SOC varies from about 48 to 51 t_C ha^{-1} (Fig. 5.9). The higher the clay content, the higher the SOC accumulation. During the

Table 5.4 Scenarios for soil heterogeneity based on different clay content

Scenarios	% of Clay
S1	3.70
S2	4.20
SBase	4.95
S3	7.60
S4	10.66

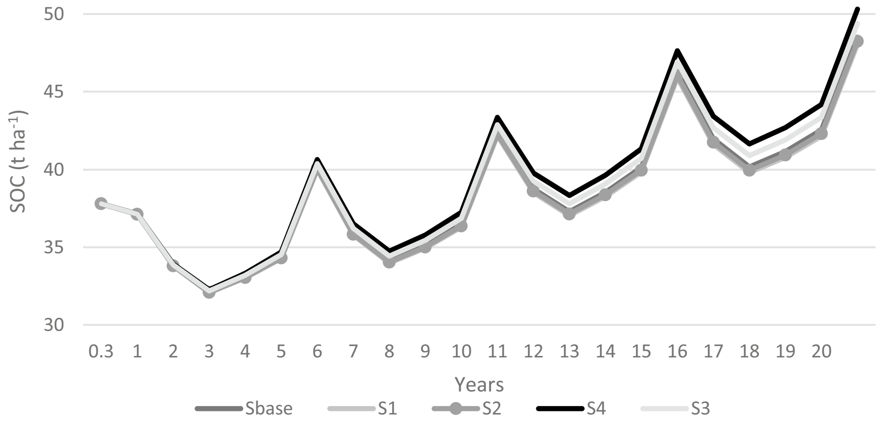


Fig. 5.9 Predicted relationship between clay content and SOC stocks

first six years, the impact of different clay content can be considered to be indifferent, as the differences between the scenarios are minimal. However, after the seventh year, the scenarios with the higher clay content (S3 and S4) start to show greater SOC levels. The impact of different clay contents is reflected in the total carbon balance of the system, which varies between −37.64 and −45.89 t_{CO2eq} ha^{-1}, indicating that the system acts as a carbon sink.

5.3.4 Influence of Aboveground Wood Production

To understand the influence of different yield amounts on SOC, the sensitivity analysis varied the aboveground wood production (AGWP) between the annual averages of 5 (Y1), 8 (Ybase), and 10 (Y3) dry tons. The results presented in Figures SM 5.1 and SM 5.2 (electronic supplementary material) show a minimal difference between the studied scenarios. For example, scenarios Y1 and Y3, which represent the 5 and 10 dry ton $year^{-1}$ scenarios, result in a relative difference of the SOC of 0.003%.

5.4 Discussion

5.4.1 Plantation Lifetime and Consideration of Wood-Carbon in the C-Balance

The analysis of the SRP revealed that its cultivation within a 20-year period can potentially mitigate climate change, as it presents a net SOC sock (Fig. 5.7). Previous studies on SRP have indicated similar results. For instance, Mishra et al. (2013) showed how the conversion from cropland to SRC-based miscanthus presented an SOC rate of 0.16–0.82 t_C ha^{-1} a^{-1}. Besides the benefits during the plantation lifetime, Whittaker et al. (2016) discussed how the end of life of the plantation, specifically, the land rehabilitation steps to recultivate the land to other land use, could disturb the SOC levels and reverse the CM benefits gained. Therefore, the carbon sequestration effect could be temporary and is dependent on the impact of the recultivation method used to terminate the plantation, as well as the subsequent land use applied. Contrary, Wachendroft et al. (2017) reported that the SOC accumulation in coarse SRP could last even after years of termination, mainly when stumps and roots that are broken down are left in the field. This emphasizes the importance of including the impacts of land transformation on SOC stocks accounting. Furthermore, it is essential that management plans also consider the factors that could reduce the CM effect. For example, regarding management plans, Rowe et al. (2020) mentioned the post-removal land management and the longevity of the SRC crop prior to reversion and soil type. Such discussion shows how considering the plantations lifetime already at the early stages of the project development supports the knowledge of the potential climate mitigation benefits of SRC plantations. For example, by knowing the potential SOC accumulation, the carbon payback period, which indicates the minimum years that are needed for sufficient SOC to accumulate and overcome emissions from agricultural practices and the generation of related products, could be estimated (Jonker et al. 2014).

5.4.2 Soil Conditions and SOC

The heterogeneity in land conditions was represented by analyzing the influence of different clay content values on SOC stocks. The analysis presented that soils with higher clay content result in greater SOC accumulation, indicating that clay content is an important factor driving the CM potential of SRP. Moreover, the results indicate the importance of understanding and accounting for heterogeneity in soil conditions. Especially, for SRP that extend to large plantation areas where geographical variation and greater heterogeneity in soil conditions are likely, accounting for such heterogeneities is of most importance. Similar findings have been recorded by Agostini et al. (2015), in which they discuss the effect of clay content on carbon retention and storage in soil, with an emphasis on its role in long-term carbon retention. The influence of clay content has been previously mentioned; for example, Jagadamma and Lal (2010) reported that the clay fraction of agricultural soils accumulated more SOC than other fractions (e.g., sand and silt fraction). Adding to this knowledge, Zhong et al. (2018) mention that the relationship between SOC and clay content is strongly influenced by climate conditions, particularly due to moisture conditions.

Besides clay content, another relevant factor influencing the net CM effect is the effect of initial SOC stocks, which are also dependent on the previous land use (Hillier et al. 2009). Parting from this knowledge, it is estimated that if for our case study the initial SOC stocks were below 34 t C ha^{-1}, combined with low clay content and a short project lifetime of 5 years, the net carbon balance would tend to be positive, thus reducing the CM effect.

5.4.3 Aboveground Carbon and SOC

There is a minimal influence of AGWP on SOC stocks (Fig. 5.9), as Peterson and Lajtha (2013) uncovered. The authors expected a positive relationship between AGWP and SOC due to the link between AGWP and leaf fall to carbon input. However, no correlation to SOC stocks, C content of the soil, or the dissolved organic carbon pool was found. An additional explanation of this result is the influence of the assumptions carried in the calculations of carbon plant input. First, it was assumed that the total aboveground wood and branches were fully collected during the harvest process; thus, the only carbon input was generated from leaf carbon input (Eq. 5.1). For estimating the leaf carbon input, the correlations presented in Fig. 5.4 show the relationship between the variables: tree age, tree height, leaf area index, and specific leaf area. However, it was not possible to establish a relationship between the previous variables and aboveground wood production (yield). Hence, it was assumed that for the scenarios (Y1, Y2, and Y3), the leaf carbon input remained constant as in the initial base case. Second, the aboveground wood production influences the carbon input through the root system (Eq. 5.2). Though, the effects due to the increase or decrease in wood yield to the root system were minimal. Therefore, the total carbon input from the scenarios studied (Y1, Y2 and Y3) had little influence on the total SOC accumulated. Similar conclusions were discussed by Hillier et al. (2009), who highlighted that variations in SOC for SRP were mainly due to the calibration of plant carbon input to soil vs yield, rather than only production yields, thereby emphasizing the need for calibration based on primary data. In conclusion, to better understand the influence of aboveground wood production, it would be necessary to deal with the assumptions described above.

5.4.4 Data and SOC Calculation

The scarcity of data is one of the biggest challenges in determining the potential CM effects of SRC-based projects. The influence of spatial heterogeneity, as clay content, indicates that SOC should be estimated at several locations within the SRP, particularly, for projects with large plantation areas. Thus, agreeing with Kalita et al. (2021), the estimation of SOC stocks should not be transferable between projects in, for instance, different geographical regions.

5.5 Conclusion

The importance of estimating the potential SOC accumulation is demonstrated by its influence on the total carbon balance of SRP production systems. Particularly, as mentioned in the Introduction, the estimation of SOC levels ex-ante or during early stages of the plantation establishment and its potential evolution during the plantation's lifetime can serve to design strategies that aid in achieving a climate mitigation effect. For instance, through the case study and sensitivity analysis in this study, the following suggestions are derived:

(i) In terms of the management plan for SRC plantations, it is suggested that during the conversion from previous land use to SRP, soil disturbances that have a negative impact on SOC stocks should be reduced (e.g., soil tillage);
(ii) Plantation maintenance, such as weed control, should be carried out by methods that have a low negative or even a positive impact on SOC stocks, for instance, manual instead of mechanical weed control;
(iii) Regarding the data collection for predicting SOC levels, decision-makers should develop a monitoring plan at the early stages of the project that involves the collection of yield, soil, climate data. Such data would serve to run the first modeling of SOC dynamics, as well as update the model together with the plantation development. Consequently, a feedback loop between data input and SOC modeling is elaborated;
(iv) Knowing the initial SOC from previous land use is essential for predicting the potential CM effect. Thus, within the project management plan, representative soil samples should be taken before the plantation's establishment;
(v) After predicting the potential development of SOC, the payback period necessary for the project to have a climate mitigation effect should be estimated.
(vi) In order to estimate accurately the potential carbon mitigation effect of SRC projects, it is necessary to include the impacts of the plantation's end of life.

In conclusion, integrating the prediction of SOC stocks into the early development stages of a SRP-based project can help project managers understand the potential CM benefits of the project, and also support the planning of sustainable management strategies that improve the CM effect. Data on SOC generated ex-ante are expected to play an important role for evaluation and decision-making including environmental considerations of investments This provides arguments for establishing such a plantation on lands where the SOC stocks could be improved by SRP. The present study has highlighted the relationship between plantation lifetime, clay content, aboveground biomass, and SOC accumulation. Thus, it shows the importance of designing SRP projects that include consistent evaluation of SOC stocks from the very beginning of the project development, as this would determine the potential CM benefits.

Acknowledgements This work received funding by the Bio Based Industries Joint Undertaking under the European Union's Horizon 2020 research and innovation program under grant agreement number 745874 and the Austria Research Promotion Agency (FFG) under the COMET program grant number 865905.

References

Albers A, Avadí A, Benoist A, Collet P, Hélias A (2020) Modelling dynamic soil organic carbon flows of annual and perennial energy crops to inform energy-transport policy scenarios in France. Sci Total Environ 718:135278. https://doi.org/10.1016/j.scitotenv.2019.135278

Agostini F, Andrew S, Goetz G, Richter M (2015) Carbon sequestration by perennial energy crops: is the jury still out? BioEnergy Res 8(3):1057–1080. https://doi.org/10.1007/s12155-014-9571-0

Barancikova G, Halás J, Gutteková M, Makovníková J, Nováková M, Skalský R, Tarasovičová Z (2010) Application of RothC model to predict soil organic carbon stock on agricultural soils of Slovakia. Soil Water Res 5:1–9. https://doi.org/10.17221/23/2009-swr

Berhongaray G, Verlinden MS, Broeckx LS, Janssens IA, Ceulemans R (2017) Soil carbon and belowground carbon balance of a short-rotation coppice: assessments from three different approaches. GCB Bioenerg 9:299–313. https://doi.org/10.1111/gcbb.12369

Buchholz T, Volk TA, Tennigkeit T, Da Silva IP (2005) Designing decentralized small-scale bioenergy systems based on short rotation coppice for rural poverty alleviation. 2–5

Clarke R, Sosa A, Murphy F (2019) Spatial and life cycle assessment of bioenergy-driven land-use changes in Ireland. Sci Total Environ 664:262–275. https://doi.org/10.1016/j.scitotenv.2019.01.397

Coleman K, Jenkinson DS (2014) RothC—a model for the turnover of carbon in soil

Don A, Osborne B, Hastings A, Skiba U, Carter MS, Drewer J, Flessa H, Freibauer A, Hyvönen N, Jones MB, Lanigan GJ, Mander Ü, Monti A, Djomo SN, Valentine J, Walter K, Zegada-Lizarazu W, Zenone T (2012) Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon. GCB Bioenerg 4:372–391. https://doi.org/10.1111/j.1757-1707.2011.01116.x

Ericsson N (2015) Time-dependent climate impact of short rotation coppice willow-based systems for electricity and heat production. Acta Universitatis Agriculturae Sueciae. Swedish University of Agricultural Sciences. https://doi.org/10.13140/RG.2.1.3508.0164

Fantin V, Buscaroli A, Buttol P, Novelli E, Soldati C, Zannoni D, Zucchi G, Righi S (2022) The RothC model to complement life cycle analyses: a case study of an Italian olive grove. Sustainability (Switz) 14. https://doi.org/10.3390/su14010569

Fontenla-Razzetto G, Wahren FT, Heilig D et al (2022) Water use of hybrid poplar (Populus deltoides Bart. ex Marsh × P. nigra L. 'AF2') growing across contrasting site and groundwater conditions in western Slovakia. Bioenerg Res

Fürtner D, Perdomo Echenique EA, Hörtenhuber SJ, Schwarzbauer P, Hesser F (2022) Beyond monetary cost-benefit analyses: combining economic, environmental and social analyses of short rotation coppice poplar production in Slovakia. Forests 13(2):349

Goglio P, Smith WN, Grant BB, Desjardins RL, McConkey BG, Campbell CA, Nemecek T (2015) Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J Clean Prod 104:23–39. https://doi.org/10.1016/j.jclepro.2015.05.040

Grogan P and Matthews R (2002) A modelling analysis of the potential for soil carbon sequestration under short rotation coppice willow bioenergy plantations. Soil Use Manage 18(3):175–183. https://doi.org/10.1111/j.1475-2743.2002.tb00237.x

Harris ZM, Spake R, Taylor G (2015) Land use change to bioenergy: a meta-analysis of soil carbon and GHG emissions. Biomass Bioenerg 82:27–39. https://doi.org/10.1016/j.biombioe.2015.05.008

Heilig D, Heil B, Kovács G, Veperdi (2020) Deliverable 2.3 field trials, growth inventory

Hillier J, Whittaker C, Dailey G, Aylott M, Cassela E, Richter GOETZM, Riche A, Murphy R, Taylor G, Smith P (2009) Greenhouse gas emissions from four bioenergy crops in England and Wales: integrating spatial estimates of yield and soil carbon balance in life cycle analyses. GCB Bioenerg 1:267–281. https://doi.org/10.1111/j.1757-1707.2009.01021.x

IPCC (2019) Refinement to the 2006 IPCC guidelines for national greenhouse gas inventories. In: Generic methodologies applicable to multiple land (Chapter 2), pp 1–96

IPCC (2022) Climate change 2022: impacts, adaptation, and vulnerability. In: Pörtner H-O, Roberts DC, Tignor M, Poloczanska ES, Mintenbeck K, Alegría A, Craig M, Langsdorf S, Löschke S, Möller V, Okem A, Rama B (eds) Contribution of Working Group II to the sixth assessment report of the intergovernmental panel on climate change. Cambridge University Press. In Press

Jagadamma S, Lal R (2010) Distribution of organic carbon in physical fractions of soils as affected by agricultural management. Biol Fertil Soils 46:543–554. https://doi.org/10.1007/s00374-010-0459-7

Jonker JGG, Junginger M, Faaij A (2014) Carbon payback period and carbon offset parity point of wood pellet production in the South-eastern United States. GCB Bioenerg 6:371–389. https://doi.org/10.1111/gcbb.12056

Kalita S, Potter HK, Weih M, Baum C, Nordberg Å, Hansson PA (2021) Soil carbon modelling in salix biomass plantations: variety determines carbon sequestration and climate impacts. Forests 12. https://doi.org/10.3390/f12111529

Lockwell J, Guidi W, Labrecque M (2012) Soil carbon sequestration potential of willows in short-rotation coppice established on abandoned farm lands. Plant Soil 360:299–318. https://doi.org/10.1007/s11104-012-1251-2

McClean GJ (2014) The impact of land-use change for lignocellulosic biomass crop production on soil organic carbon stocks in Britain. The University of Edinburgh

Meyer M, Morgenstern K, Heilig D, Heil B, Kovács G, Leibing C, Krabel D (2021) Biomass allocation and root characteristics of early-stage poplars (populus spp.) for assessing their water-deficit response during SRC establishment abstract. BioEnergy Res 14(2):385–398. https://doi.org/10.1007/s12155-021-10264-6

Minasny B, Malone BP, McBratney AB, Angers DA, Arrouays D, Chambers A, Chaplot V, Chen Z, Cheng K, Das BS (2017) Soil carbon 4 per mille. Geoderma 59–86

Mishra U, Torn MS, Fingerman K (2013) Miscanthus biomass productivity within US croplands and its potential impact on soil organic carbon. GCB Bioenerg 5:391–399. https://doi.org/10.1111/j.1757-1707.2012.01201.x

Morais TG, Silva C, Jebari A, Álvaro-Fuentes J, Domingos T, Teixeira RF (2018) A proposal for using process-based soil models for land use life cycle impact assessment: application to Alentejo, Portugal. J Clean Prod 192:864–876

Pacaldo RS, Volk TA, Briggs RD (2013) No significant differences in soil organic carbon contents along a chronosequence of shrub willow biomass crop fields. Biomass Bioenerg 58:136–142. https://doi.org/10.1016/j.biombioe.2013.10.018

Perdomo EEA, Brunnhuber N, Hesser F (2021) Deliverable 5.5 integration of LCA in value chain establishment: preliminary results. https://cordis.europa.eu/project/id/745874/results

Petersen BM, Knudsen MT, Hermansen JE, Halberg N (2013) An approach to include soil carbon changes in life cycle assessments. J Clean Prod 52:217–224. https://doi.org/10.1016/j.jclepro.2013.03.007

Peterson FS, Lajtha KJ (2013) Linking aboveground net primary productivity to soil carbon and dissolved organic carbon in complex terrain. J Geophys Res Biogeosci 118:1225–1236. https://doi.org/10.1002/jgrg.20097

Rampazzo Todorovic G, Stemmer M, Tatzber M, Katzlberger C, Spiegel H, Zehetner F, Gerzabek MH (2010) Soil-carbon turnover under different crop management: evaluation of RothC-model predictions under Pannonian climate conditions. J Plant Nutr Soil Sci 173:662–670. https://doi.org/10.1002/jpln.200800311

Rossi V (2018) IKEA LUC guidance pilot: methodological points of the Slovakia case studies

Rowe RL, Keith AM, Elias DMO, McNamara NP (2020) Soil carbon stock impacts following reversion of Miscanthus × giganteus and short rotation coppice willow commercial plantations into arable cropping. GCB Bioenerg 12:680–693. https://doi.org/10.1111/gcbb.12718

Rumpel C, Amiraslani F, Chenu C, Garcia Cardenas M, Kaonga M, Koutika LS, Ladha J, Madari B, Shirato Y, Smith P, Soudi B, Soussana JF, Whitehead D, Wollenberg E (2020) The 4p1000 initiative: opportunities, limitations and challenges for implementing soil organic carbon sequestration as a sustainable development strategy. Ambio 49:350–360. https://doi.org/10.1007/s13280-019-01165-2

Rytter R-M, Rytter L, Högbom L (2015) Carbon sequestration in willow (Salix spp.) plantations on former arable land estimated by repeated field sampling and C budget calculation. Biomass Bioenerg. https://doi.org/10.1016/j.biombioe.2015.10.009

Schmidt-Walter W (2019) Evaluation of environmental impacts of short rotation coppice with regard to the amount and quality of groundwater recharge. Georg-August-University Göttingen

Wachendorf C, Stuelpnagel R and Wachendorf M (2017) Influence of land use and tillage depth on dynamics of soil microbial properties soil carbon fractions and crop yield after conversion of short-rotation coppices. Soil Use Manage 33(2):379–388. https://doi.org/10.1111/sum.2017.33.issue-2, https://doi.org/10.1111/sum.12348

Whittaker C, Macalpine W, Yates NE, Shield I (2016) Dry matter losses and methane emissions during wood chip storage: the impact on full life cycle greenhouse gas savings of short rotation coppice willow for heat. BioEnerg Res 9:820–835. https://doi.org/10.1007/s12155-016-9728-0

Zanchi G, Frieden D, Pucker J, Bird DN, Buchholz T, Windhorst K (2013) Climate benefits from alternative energy uses of biomass plantations in Uganda. Biomass Bioenerg 59:128–136. https://doi.org/10.1016/j.biombioe.2012.03.023

Zhong Z, Chen Z, Xu Y, Ren C, Yang G, Han X, Ren G, Feng Y (2018) Relationship between soil organic carbon stocks and clay content under different climatic conditions in Central China. Forests 9. https://doi.org/10.3390/f9100598

Chapter 6
Assessment of Greenhouse Gas Emission Reduction from Biogas Supply Chains in Germany in Context of a Newly Implemented Sustainability Certification

Nora Lange, David Moosmann, Stefan Majer, Kathleen Meisel, Katja Oehmichen, Stefan Rauh, and Daniela Thrän

Abstract Life cycle assessments (LCA) approaches, analysing potential impacts associated with the production and use of biomass for energy and material purposes, have become increasingly important in recent years. An internal project at the Deutsches Biomasseforschungszentrum- DBFZ is investigating, which priority areas have been addressed at the institute with LCA. The preliminary results of the study show mostly practice-linked applications with a focus on the assessment of fuels, their production and technical feasibility. In this publication, we present one of the studies analysed, in which a simplified LCA approach defined in the renewable energy directive (RED II), was applied. Based on primary data from 10 biogas and biomethane supply chains in Germany, the applicability of the RED II greenhouse gas (GHG) emission calculation approach was analysed. Most of the biogas plants assessed were found to be compliant with the required minimum GHG emissions reduction. Storage of digestate, N-fertilization and the use of fossil diesel were identified as the main factors, influencing the GHG intensity of the respective value chains. Additionally, individual calculation requires a high effort for data collection. The availability of tools and default values could therefore support market actors with an efficient implementation of the RED II.

Keywords Bioenergy · RED II GHG calculation methodology · Climate impact · Sustainability certification · Biogas supply chains

N. Lange (✉) · D. Moosmann · S. Majer · K. Meisel · K. Oehmichen · D. Thrän
Deutsches Biomasseforschungszentrum Gemeinnützige GmbH, Torgauer Straße 116, 04347 Leipzig, Germany
e-mail: Nora.Lange@dbfz.de

S. Rauh
Fachverband Biogas E.V, Angerbrunnenstraße 12, 85356 Freising, Germany

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*,
Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_6

6.1 Background

The use of the Life Cycle Assessment (LCA) approach, which can be used to assess potential impacts associated with the life cycles of biomass for energy and material purposes, has become an increasingly important instrument in science and also in political decisions (McManus and Taylor 2015; Sala et al. 2021). Acknowledging the complexity of value chains for biobased products and the variety of research questions from different stakeholder groups, several adaptations of the LCA approach have been developed, allowing assessments on the product, process, company or project level within the framework of the bioeconomy. In order to analyse this development and to prepare the implementation of new approaches and tools for the sustainability assessment of biobased value chains, the Deutsches Biomasseforschungszentrum (DBFZ) has launched an internal project, analysing more than 85 assessment projects, which have been carried out in the DBFZ since 2008.

For this internal project, the inventory data of the various LCA projects were structured and analysed regarding a number of defined criteria (e.g., use and origin of substrates and feedstocks, temporal aspects of the projects, technical background, process focus as well as sector of application, etc.). As an example, Fig. 6.1 shows the distributions of the energy sectors and energy carriers, which have been in the focus of interest in the studies analysed. It can be seen that biofuels for transportation purposes play the most important role, especially biodiesel, bioethanol and biomethane. Biogas and biomethane can be used to supply energy in various sectors. While most biogas plants produce electricity, the upgrading of biogas and the subsequent distribution of biomethane in a gas grid is a promising alternative for operators. Biomethane is considered an interesting bioenergy option, due to its high flexibility and, the potentially high GHG savings, especially from the use of wastes and residues (Bundesanstalt für Landwirtschaft und Ernährung 2021; Wietschel et al. 2019) (Fig. 6.1, right side).

One of the DBFZ focus areas of applied research is on the issues of Renewable Energy Directive (RED) and its implementation/impact on practice. Thus, in this article we present a recently completed project. It is one of the 85 projects evaluated in the internal project and deals with the new certification requirements on the biogas and biomethane market according to RED II.

6.2 Introduction

One example of the practical application of the general LCA approach is the GHG emission calculation within the context of the EU RED II. The overall goal of this directive is to promote energy from renewable sources. In the case of energy production from biomass, the directive includes, amongst others, sustainability criteria and criteria for GHG emission savings. Compliance with the criteria is a precondition for public support as well as the consideration of the respective biomass for the fulfilment

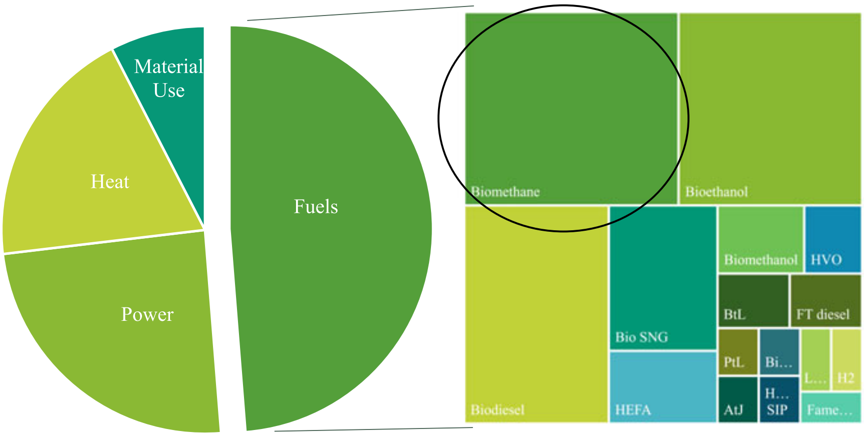


Fig. 6.1 Figure adapted from (Lange et al. 2020). Sectors investigated with LCA in DBFZ projects 2008–2022 (left) and fuels analysed in detail (right); biomethane was at the top of all fuels examined. Explanation of used abbreviations: Bio SNG—synthetic natural gas, HEFA—hydroprocessed esters and fatty acids, HVO—hydrotreated vegetable oils, BtL—biomass to liquid, FT diesel—fischer tropsch diesel, PtL—power to liquid, AtJ—alcohol to jet, HFS-SIP—hydroprocessing of fermented sugars—synthetic iso-paraffinic kerosene, LNG—liquefied natural gas, H_2—hydrogen, Fame—fatty acid methyl esters/Fage—fatty acid glycerol formal ester

of the national targets for the development of renewable energy, as defined in energy and climate policies. Operators within the scope of the RED II can prove compliance with the sustainability requirements, based on a certificate from a recognized certification scheme. The directive and the respective sustainability requirements have been introduced, firstly for liquid biofuels in 2009. The revised RED (RED II) was published in 2018 and became effective in 2021 (European Commission 2018). This revision included an extension of the scope to electricity, heating and cooling from solid and gaseous biomass fuels used in installations above a fixed capacity threshold, which is 2 MW in the case of gaseous biomass fuels. Annexes of the directive specify GHG emission calculation rules and curtail the goal and scope. The approach can therefore be regarded as a simplified LCA based assessment approach. More comprehensive approaches, in accordance with ISO 14040/14044 for instance, allow for higher degrees of freedom and flexibility regarding certain methodological choices such as for example impact categories, characterization factors, system boundaries, allocation approaches and more. This is why many LCA studies are often consistent in the methodology, but the results are not directly comparable (Roßmann et al. 2019). However, depending on the application, more comprehensive approaches may be favourable for some applications, for example, to investigate extensive research questions. For regulation purposes or within a certification context, more simple and robust approaches seem more manageable and therefore preferable. In case of the RED II approach, additional guidance and supportive elements, such as default values shall support the operationalisation and feasibility of GHG emission calculations in practice, allowing for a calculation approach which also allows for a direct comparison of the GHG mitigation potential of different biofuel options.

Compliance with the GHG emissions saving criteria of the RED II is shown based on a life cycle approach, meaning that emissions along the entire life cycle of energy from biomass or waste flows are considered. This includes feedstock cultivation or collection, transport, distribution, processing and energy generation. Potential savings are calculated based on comparison to a defined reference value ("fossil fuel comparator" in the terminology of the RED II). In case of electricity from biogas, a GHG emission reduction of at least 70% has to be proven. This applies to installations starting operation as of 2021. This threshold will increase to 80% for installations with a starting date in 2026 or afterwards (European Commission 2021). For defined value chains, respective default values in RED II may be used to reduce the administrative burden for operators in the certification process. This option has been widely applied in the past. Meanwhile, since the market conditions in some energy sectors might allow for price premiums for biofuels with comparably high GHG mitigation potentials, the demand for individual calculations has significantly increased. This is at least the case in Germany, where the system for biofuel blending has originally been based on energy-based targets and was replaced by a GHG mitigation quota in 2015 (Naumann et al. 2021). As a result, there can be market preference for biofuels with higher GHG emissions saving over fuels with less positive GHG emission intensities, dependent on the production costs. For energy production from biogas, which is the focus of this study, default values are available for maize, biowaste and manure only (European Commission 2018). However, substrate mixtures in biogas plants often include a variety of different feedstocks. In addition to the ones mentioned, grass silage, grains and whole-plant silage from different grains, catch crops and sugar beet are further feedstocks, which are currently mainly used in Germany (Daniel-Gromke et al. 2017). For these reasons, the calculation of so-called actual values, in the sense of individual emission calculations, is expected to become more and more relevant.

Different studies have been conducted focussing on the GHG emission calculation for bioenergy concepts in the context of the Renewable Energy Directives. Some of them assessed the methodological approach itself (Börjesson et al. 2015; Czyrnek-Delêtre et al. 2017; Manninen et al. 2013; Whittaker 2015). Others conducted scenario based assessments and focused on specific value chains, feedstocks or regional characteristics, e.g. the production of biomethane from grass (Rasi et al. 2020; Smyth et al. 2009) or biomethane from willow used for heat and transport in Ireland (Long et al. 2021). Rana et al. investigated four different biogas feedstocks in an electricity supply chain in southern Italy based on the legal policy framework in 2015 (Rana et al. 2016). A more recent study from Finland focussed on digestate as co-product from anaerobic fermentation and analysed different methods for co-product allocation (Timonen et al. 2019).

This study adds to this existing inventory, as we analysed the processes concerning requirements on GHG emission mitigation in a certification context, with a specific focus on the perspective of economic operators of bioenergy production. Although the RED II includes the core calculation principles, application in practice can be difficult because of the abstract and concise way in which they are presented. We therefore developed a biogas- and biomethane-specific GHG emission calculation methodology, aiming at complementing the RED II and designed to support stakeholders within the certification context. Based on this methodology, 10 assessments, for selected existing biogas supply chains, which represent typical cases for German conditions, were examined. Input data for the calculations was obtained from the selected biogas facilities located in different regions across Germany.

Based on the results of the assessments, the following two research questions could be answered:

1. Which are the decisive factors affecting the compliance with the RED II GHG reduction requirements of typical biogas supply chains in Germany?
2. Is the process to proof compliance with the RED II feasible for economic operators in the scope, based on the available regulations, guidelines and tools?

6.3 Methodological Approach

6.3.1 Method

The assessment of GHG emissions for the 10 biogas plants has been based on the general requirements of the RED II. The principal scope and calculation rules for GHG emission reduction from biogas and biomethane value chains are included in Annex V and VI of the RED II (European Commission 2018) and are compiled in this section. With respect to the system boundaries, there is a slight difference between heat and power from biogas and biomethane for transport. In the former, system boundaries are considered well-to-grid, while the latter can be considered well-to-tank. Following the definitions in the RED II, emissions from fuel in use (combustion in the engine) are set to zero for biofuels. However, non-CO_2 GHG emissions are taken into account for the use of biogas for electricity production. The general rationale behind this approach is the simplified assumption, that the amount of the biogenic CO_2 emitted when biogas is combusted equals the amount of CO_2 absorbed during biomass growth. The three GHGs carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are to be considered. Conversion factors to calculate CO_2-equivalents (CO_2eq) are 1, 25 and 298 respectively. In case of co-digestion of different substrates, which is the predominant biogas process concept in Germany, emissions from the supply chain interfaces up to the feedstock processing are calculated for each feedstock individually (cp. Eq. 6.1). A factor expressing the share of feedstock in the feedstock mix (S_n), in terms of the energy content, provides for the contribution of emissions from the single feedstock to the mix. Emissions from feedstock cultivation and extraction (e_{ec}), transport and distribution (e_{td}), (direct) land use change (e_l), processing (e_p) and fuel in use emissions (e_u) contribute to the total emission value. Negative emissions from carbon capture and storage (e_{ccs}), carbon capture and replacement (e_{ccr}) as well as from soil carbon accumulation via improved agricultural practices (e_{sca}) can lower the total emission intensity. Moreover, in the latter term, a bonus[1] for emission avoidance via manure digestion can be added.

$$E = \sum_{1}^{n} S_n \times \left(e_{ec,n} + e_{td,feedstock,n} + e_{l,n} - e_{sca,n}\right) + e_p + e_{td,product} + e_u - e_{ccs} - e_{ccr} \quad (6.1)$$

The calculation of emissions from biomass cultivation (e_{ec}) includes nitrous oxides (N_2O) emissions from nitrogen (N) application. A draft implementing regulation includes a specification (European Commission 2021). Due to reasons we discuss in this article, we applied a simplified calculation approach described in (Rauh 2010) where needed. This approach is similar to the IPCC Tier 1 approach (IPCC 2006), but limited to direct N_2O emissions and does not include plant and site specific parameters. Total emissions are to be calculated per 1 MJ biofuel or final energy (functional unit), respectively. This enables the calculation of emissions saving according to formula (6.3) considering the fossil fuel comparators[2] (EC_F) given in Annex V and Annex VI of RED II. For combined heat and power processes, total emissions (E) need to be allocated to heat (EC_h) and electric power (EC_{el}) beforehand. This is done by means of exergy allocation according to formula 6.2 (electricity) considering the exergy content of heat (C_h) and electricity (C_{el}) and the thermal (η_{th}) and electrical efficiency (η_{el}).

$$EC_{el} = \frac{E}{\eta_{el}}\left(\frac{C_{el} \times \eta_{el}}{C_{el} \times \eta_{el} + C_h \times \eta_h}\right) \quad (6.2)$$

$$GHG\ Emissions\ Saving = \frac{EC_F - EC_B}{EC_F} \times 100 \quad (6.3)$$

6.3.2 General Approach and Dataset

As part of the overall approach (Fig. 6.2), the first step was the development of a process for the selection of a number of biogas and biomethane plants to be included in the assessment. This was achieved with the help of a questionnaire, which was distributed among biogas plant operators, identifying operators with a general interest for participation. Based on the completed questionnaires, a preselection of 37 interested operators was compiled. Building on the information gained

---

[1] A bonus of 45 g CO_2eq/MJ manure (– 54 kg CO_2eq/t fresh matter) accommodates for methane emissions during manure storage, which are avoided when manure is used in a biogas digester.

[2] 94 g CO_2eq/MJ (Biofuels), 183 g CO_2eq/MJ (electricity), 80 g CO_2eq/MJ (heat).

with the questionnaires, the 37 biogas plants were characterized by means of the following parameters:

- Federal state
- Commissioning date
- Type of energy generation: on-site CHP, biomethane CHP, biomethane transport fuel
- Plant capacity (installed capacity, rated capacity, upgrading capacity)
- Feedstocks (waste and residues, cultivated biomass)

The characterisation of the preselected biogas plants enabled a subsequent step, in which we selected 10 facilities covering a wide spectrum of capacities, feedstock and other parameters (Table 6.1). This (final) selection served as sample for the assessment. None of the biogas plants applied a carbon capture technology. Moreover, with respect to the use of cultivated biomass, no direct land use change was reported. In the calculations, the terms e_{ccs}, e_{ccr} and e_l in formula (6.1) were thus set to zero.

In line with the draft calculation methodology, an excel tool was developed to conduct the calculations and to support the collection of primary data. This tool combined the collected primary data, the relevant comparators for fossil fuels as well as the calculation formulas (6.3 (1)–(3)) and the relevant emission factors. To compensate for gaps in the collected data, some standard calculations values were used in the calculations (Table 6.2).

Calculations were reviewed and verified by a recognized certification body. The verification reports were incorporated into an iterative process for further development of the methodology draft and consequently the supporting tool with the aim to increase its robustness, RED II conformity and practical relevance of the calculations (Fig. 6.2).

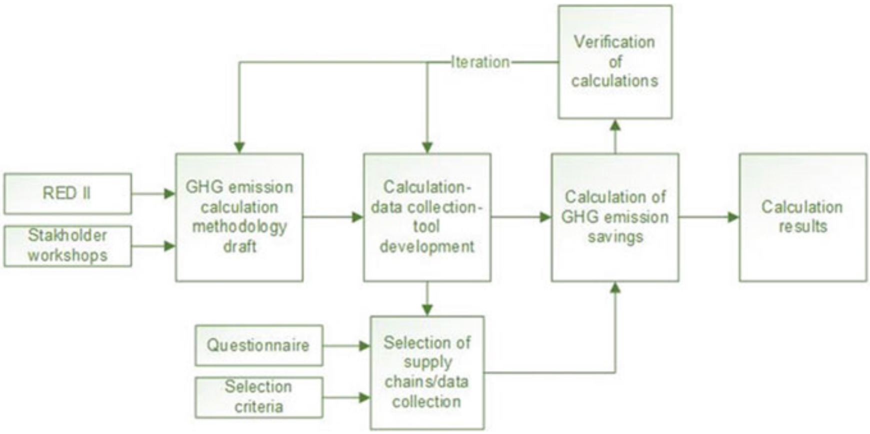


Fig. 6.2 Stepwise overall methodological approach of the ZertGas project to calculate GHG emissions savings in typical biogas and biomethane supply chains

Table 6.1 Basic process data and feedstock composition of participating biogas plants. Collection of data was done in 2021

Operator ID	Capacity, kW	Digestate storage	Biogas upgrading	Feedstock composition, %[3]		
				Cultivated biomass	Waste and residues	Manure/ slurry
1	2,575	Closed	No	83	0	17
8	760	Closed	No	0	62	38
10	305	Closed	No	92	0	8
19	2,864	Open	No	86	0	17
20	360	Open	No	81	0	19
29	330	Closed	No	0	0	100
33	3,077	Closed	Yes	79	4	17
34	3,120	Closed	No	38	2	60
35	2,864	Open	No	59	0	41
36	400	Closed	Yes	100	0	0

Table 6.2 Various standard calculation values and assumptions used in the calculations

Parameter	Value	Unit	Source
Diffuse CH_4 emissions in biogas fermenter	1	% (of methane yield)	Haenel et al. (2020); IPCC (2019); Vogt (2008)
CH_4 and N_2O emissions from open digestate storage - manure	69.56	g CO_2eq/ MJ	BioGrace (2021)
CH_4 and N_2O emissions from open digestate storage - biowaste	21.85	g CO_2eq/ MJ	BioGrace (2021)
CH_4 and N_2O emissions from open digestate storage - maize	13.51	g CO_2eq/ MJ	BioGrace (2021)
CH_4 and N_2O emissions from biogas CHP gas engine	8.92	g CO_2eq/ MJ	BioGrace (2021)

6.4 Results

6.4.1 *Major Drivers and Compliance with RED II*

The variety of the selected supply chains (e.g. in terms of plant configuration, feedstock focus etc.) (Table 6.1) is reflected in the calculation results, which reveal significant differences between the analysed plants (Fig. 6.3). Fuel-in use emissions (e_u) are

[3] Based on energy content.

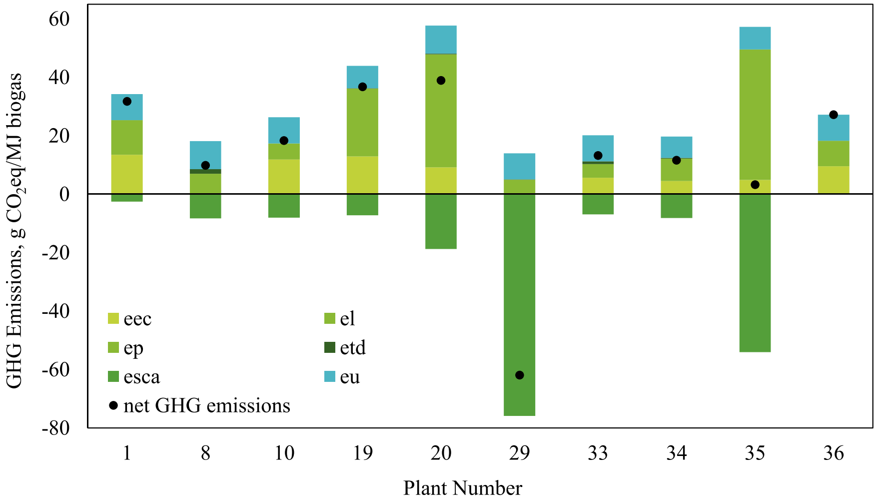


Fig. 6.3 Total GHG emissions per energetic unit of biogas of the assessed supply chains and contributions of the individual terms of formula (6.1) (e_{ec} = emissions from cultivation and extraction, e_l = emissions from land use change, e_p = processing emissions, e_{td} = emissions from transport and distribution, e_u = fuel in use emissions, e_{sca} = emission savings from soil carbon accumulation via improved agricultural management)

based on standard calculation values (Table 6.2), as in none of the CHPs non-CO_2-emissions were measured or continuously monitored. For that reason, this contribution does not impact the variety of the total emissions. The feedstock transport (e_{td}) did not influence the result largely. The highest contribution was determined for plant no. 8 with 1,6 g CO_2eq/MJ, which can be explained with a high share of food waste (62%) in the feedstock mix. However, some operators did not report distinguished data for e_{td}, which is therefore zero in the particular bars in Fig. 6.3. In these cases, emissions from diesel consumption during transport are included in e_{ec}.

Across all analysed production plants, especially the process of the feedstock processing (e_p) showed great differences. The magnitude of emissions accounted for in this term is firstly affected by the type and demand of electrical and thermal process energy (internal or external supply) as well as the methane slip in the digester. For the latter no measurements were conducted at the assessed plants and a standard assumption was used to estimate CH_4 emissions. Secondly and more relevant are emissions from digestate storage, which are considered in e_p, as well. These showed the highest impact. The processing emissions for biogas plants with open digestate storage (no. 19, 20, 35) are therefore very high when compared with plants with closed digestate storage. The term e_{sca} includes emission savings from soil carbon accumulation as well as improved manure management practices, namely emission reduction via digestion of manure. All emission savings considered in our assessments are related to the latter. Considering the feedstock composition, the results indicated a correlation between the share of manure in the mix and the CO_2 equivalents saved and thus decreasing the total emission value. Due to the bonus for manure utilization as feedstock, an entirely manure based feedstock mix enables to even achieve negative emissions, as can be seen from the result of no. 29.

When looking at the results and disregarding emissions from open digestate storage, as these are technically avoidable, the cultivation of biomass remains a relevant driver. This becomes explicitly obvious from the results of plant no. 1, 10, 19, 20, 33 and 36, as their configuration is characterized by higher ratios (around 80–100%) of cultivated biomass in the feedstock mix. Within the cultivation stage there are two major drivers –nitrogen fertilization and the consumption of fossil diesel (Fig. 6.4). The energy intensive production process of synthetic N-fertilizer is one of the reasons for the high contribution of emissions related to N-fertilization. Direct and indirect N_2O emissions caused by nitrogen application result in high GHG emissions and are the main driver within the cultivation phase. Thereby, according to the appropriate default emission factors (IPCC 2006), nitrogen from synthetic and organic fertilizers both contribute to the release of N_2O emissions.

The RED II threshold for the GHG emissions saving criteria of electricity from biogas, valid at the time we conducted the study was 70%. Three of 10 assessed plants could not comply with the criterion (Fig. 6.5). The results were in a range between 60 and 160%. Six of the plants could comply with the 80% savings requirement for future installations starting operation from 2026 onwards. The lowest savings were calculated for two plants with open digestate storage systems.

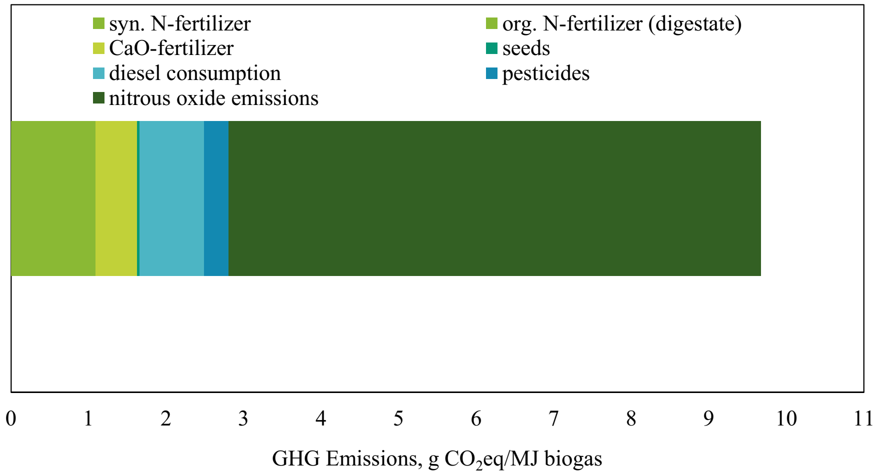


Fig. 6.4 Total emissions from maize cultivation. Own calculation based on a modeled representative production system

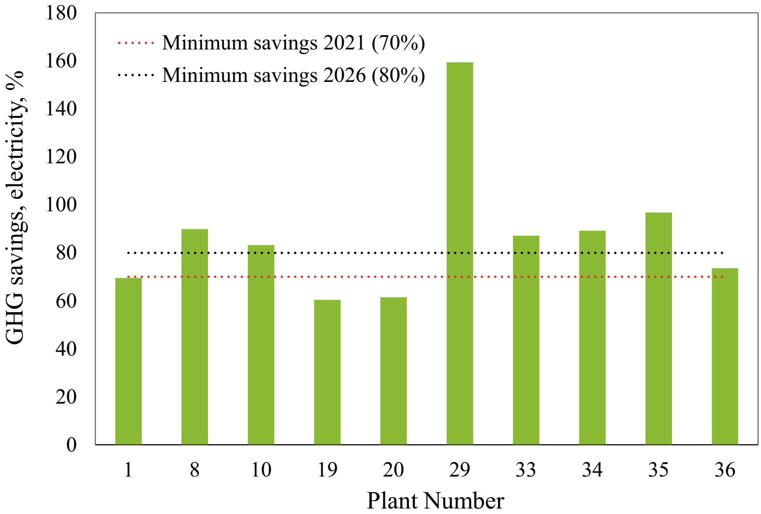


Fig. 6.5 GHG emission savings of assessed supply chains for electricity from biogas

6.4.2 General Feasibility

With respect to the development of a robust and auditable GHG emission calculation by plant operators, we found the applicability of the current GHG calculation approach of the RED II to be low. This was mainly due to the following reasons: The provision of the calculation and data collection tool was not sufficiently supporting the operators to prepare the calculation independently. Further queries, assistance and adjustments were needed to complete a calculation suitable for submission to the auditor.

Equally important, we found the availability of specific input data, emission factors and other, necessary guidance from the EU Commission incomplete. There are many optional specific additives in the biogas process. While primary data on the consumed amount of these products within the balancing period was available, corresponding emission factors could not be included for some additives in the calculation because of inaccessibility. Amongst others, this applied to micro nutrients and more specified products like iron chloride, ferrous oxide and others. Available lists with standard calculation values and emission factors are provided by some certification schemes as part of the system documentation (e.g. ISCC System GmbH 2021) as well as by the European Commission (European Commission 2015, 2021). Very specific emission factors are not included in the mentioned sources and even in more advanced LCA databases some values could not be found.

The methodological framework is determined by the Annexes of RED II and an implementing regulation specifying the former. At the time we conducted the study, the latter was published in a draft version only (European Commission 2021). However, the given information can be assumed likely to become valid, which is why we considered these regulations. In the named draft implementing regulation, the methodology for calculating nitrous oxide emissions is specified. However, we found the calculation according to this method hardly feasible in practice due to its general complexity. Moreover, additional data (e.g. soil carbon content, soil pH) is needed to calculate a crop- and site-specific emission factor.

6.5 Discussion

6.5.1 Assessment of Conformity and Potential for Optimization

The calculations of GHG emissions saving in typical biogas supply chains showed that GHG emission savings criteria could be met by the majority of assessed cases. Furthermore, the results indicated that even the raised threshold, valid as of 2026 (80%) seems generally achievable. Based on our assessment, we identified the way of digestate storage as the low hanging fruit for optimization of the total GHG emissions saving, because open digestate storage was found to be a major emission source, which is supported by findings of other studies, e.g. (Timonen et al. 2019). Closing an open storage tank is a technically available solution and already widely applied measure that offers a possibility to achieve higher GHG emission savings without conceptual changes of the supply chain. However, the required minimum GHG emissions saving can be achieved with an open digestate storage, as could be shown with the results for plant no. 35. In this plant a relatively high share of manure in the feedstock mix could compensate emissions from digestate storage. Operators willing to further decrease the total emissions should focus on the feedstock mix and increase the share of waste and residue materials in the feedstock mix. Optimization in biomass cultivation could start with questioning the nitrogen application level. Further research should assess if a loss in crop yield due to a reduction of the fertilization level could be compensated by higher GHG emission savings. Emissions from fossil diesel in farming was identified as another relevant driver, as indicated by similar results of other studies, e.g. (Rana et al. 2016). The use of biofuels allows the application of the RED II default values instead an emission factor for fossil diesel (European Commission 2021). In case of biodiesel, these emissions could thus be decreased by 47%-84%, dependent on the biodiesel feedstock (European Commission 2018). An additional option for further improvement of the GHG balance of cultivated biomass could therefore be a substitution of diesel in agricultural machinery by biodiesel or biomethane.

6.5.2 Feasibility to Proof Compliance by Operators

With respect to the feasibility, data collection procedures should be improved, specifically on the level of biomass production and provision. In four of the assessed supply chains, total diesel consumption was considered, but consumption data could not be differentiated between diesel used for farming and for feedstock transport. This might lead to non-conformities in the certification process in two ways. The verification of GHG balances is (in parts) based on a comparison with disaggregated typical values given in Annex V and VI of the RED II on the one hand. This comparison is carried out as a plausibility check. Aggregation of emission values therefore hampers the verification done by auditors and reviewers of certification bodies. On the other hand, calculated emissions have to be reported to the national authority as disaggregated emission values according to the particular terms of formula (6.1). Systematic collection of data by farmers seems therefore required. For cultivated biomass, data on the plot level was not available in all cases. We do not expect significant differences in the results, if data with this level of detail would have been included in the calculations. However, in the actual certification context, data on biomass cultivation is expected to be required on plot level. This will increase the overall complexity and feasibility for operators.

In bioenergy supply chains within the RED II scope, each interface will be certified individually and calculated GHG emissions are transmitted throughout the supply chain from one interface to the next one downstream. Thereby the GHG emission calculation is of limited effort and complexity for a single interface. In energy production from biogas, there are often many interfaces or even all interfaces of one supply chain combined (super interface). That is because biogas plants are often located on farms where parts or all of the biomass of the feedstock mix is produced, processed and converted to energy. As a result, the GHG emission calculation will be done by one operator only. Due to the variety of feedstock streams, this can lead to the challenge of collecting and processing a large amount of data. In our study this specialty occurred by design, as for practical reasons, we approached the operators only and not each interface along the supply chain. However, we argue that this is close to practice and thus our observations with respect to the feasibility within our study are also relevant for the actual certification.

Default values for emissions savings can support the certification process and their application might be preferable, especially for operators not targeting a maximum GHG emission reduction (default values are based on conservative assumptions). While liquid biofuel pathways, especially for first generation biofuels (e.g. rapeseed biodiesel or sugar beet bioethanol) are well represented in the list of default values in the Annexes of the RED II, default values for biogas value chains are very limited to pathways based on maize, biowaste and manure. From the assessed supply chains, total default values would have been applicable in two of 10 cases only due to the variety of feedstocks. This stresses the relevance of the individual GHG emission calculation in the biogas and biomethane sector under current conditions as well as the limitation of default values applicable to biogas and biomethane pathways.

However, a lot of values are needed for individual GHG emission calculation, but standard calculation values such as emission factors, NUTS2 values (regionalized values for e_{ec}) and calculation tools supporting the individual calculation are sparsely available. In that respect, we found the needs of the biogas sector to be insufficiently addressed so far. This is mainly linked to the variety of feedstocks included in the supply chains. As the implementing regulation prescribes that the inherent emission factors shall be used (European Commission 2021), this list should include more factors.

The individual calculation of N_2O emissions according to the implementing regulation was not found practicable due to its complexity. The regulation draft allows using the GNOC (Global Nitrous Oxide Calculator) calculation tool. However, the use of the tool is questionable for different reasons. Firstly, it does not include sufficient feedstocks. Secondly, it was developed for liquid biofuels. In biodiesel production, mostly parts of the plants (grain, rape seeds, etc.) are processed, it is not clear how to rate the use of substrates in the biogas supply chains, where mostly whole plants are processed. The amount of above ground residues left on the field might be different in the two cases. Hence, N_2O-Emissions would differ as well, as nitrogen in plant residues is one source of such. Consequently, the tool should be updated taking the extension of the RED II scope into account and distinguishing between feedstocks for different bioenergy pathways.

6.5.3 *Limitations of the Study*

The individual calculation of GHG emission savings was found challenging for operators. With respect the feasibility, it has to be noted, that the tool we used was not fully optimized for application in practice in terms of usability. Moreover, operators were participating in an assessment of a research project and not in an actual certification process. This might limit our evaluation of the feasibility to some extent.

6.6 Outlook

The results of the wider, internal analysis for 85 projects with an LCA focus, performed at the DBFZ until 2022 will be published in an internal project report. This step is currently under preparation. The publication will also highlight various trends in the field of LCA and bioenergy in Germany over the last 14 years, address relevant developments and discussions such as indirect land use change, social LCA or spatial/regional LCA and their scientific challenges in the coming years.

As a further step to improve internal accessibility, it is planned to transfer selected data sets into a new database format. This database could also be made accessible via a web app and later made available to external users.

Another central point of the internal project is the networking and cooperation with other institutes, scientists and the possible realisation of data collections of common knowledge. Various European projects are working together to make the instrument of LCA more widely and uniformly available across Europe. Strategies are being investigated on how to make data available to the public, whether for comprehensive sustainability assessments, as a guide for policy instruments or for educational transfer. We would like to continue promoting the important exchange of scientific data in the future and are looking for like-minded people for this project.

Acknowledgements The ZertGas project was funded under the Project Management Jülich by the Federal Ministry for Economic Affairs and Climate Action (BMWK) on the basis of a decision by the German Bundestag (funding code 03KB164A).

References

BioGrace (2021) BioGrace II calculation tool: harmonised greenhouse gas calculations for electricity, heating and cooling from biomass. www.biograce.net

Börjesson P, Prade T, Lantz M, Björnsson L (2015) Energy crop-based biogas as vehicle fuel—the impact of crop selection on energy efficiency and greenhouse gas performance. Energies 8:6033–6058. https://doi.org/10.3390/en8066033

Bundesanstalt für Landwirtschaft und Ernährung (2021) Evaluations- und Erfahrungsbericht für das Jahr 2020: Biomassestrom-Nachhaltigkeitsverordnung Biokraftstoff-Nachhaltigkeitsverordnung. https://www.ufop.de/files/5516/3966/9053/BLE_Evaluationsbericht_2020.pdf. Accessed 3 May 2022

Czyrnek-Delêtre MM, Smyth BM, Murphy JD (2017) Beyond carbon and energy: the challenge in setting guidelines for life cycle assessment of biofuel systems. Renew Energy 105:436–448. https://doi.org/10.1016/j.renene.2016.11.043

Daniel-Gromke J, Rensberg N, Denysenko V, Trommler M, Reinholz T, Völler K, Beil M, Beyrich W (2017) Anlagenbestand Biogas und Biomethan – Biogaserzeugung und -nutzung in Deutschland: DBFZ Report Nr. 30, Leipzig

European Commission (2015) Standard calculation values v.1.0. Excel Datenbank. https://ec.europa.eu/energy/topics/renewable-energy/biofuels/voluntary-schemes_en?redir=1. Accessed 15 June 2020

European Commission (2018) RICHTLINIE (EU) 2018/2001 DES EUROPÄISCHEN PARLAMENTS UND DES RATES - vom 11. Dezember 2018 - zur Förderung der Nutzung von Energie aus erneuerbaren Quellen: RED II

European Commission (2021) COMMISSION IMPLEMENTING REGULATION (EU) on rules to verify sustainability and greenhouse gas emissions saving criteria and low indirect land-use change-risk criteria (draft). https://ec.europa.eu/info/law/better-regulation/have-your-say/initiatives/12723-Sustainable-biofuels-bioliquids-and-biomass-fuels-voluntary-schemes-implementing-rules-_en. Accessed 6 March 2022

Haenel H-D, Rösemann C, Dämmgen U, Döring U, Wulf S, Eurich-Menden B, Freibauer A, Döhler H, Schreiner C, Osterburg B, Fuß R (2020) Calculations of gaseous and particulate emissions from German agriculture 1990–2018 (RMD): Report on methods and data (RMD) Submission 2020. Thünen Report, Braunschweig

IPCC (2006) 2006 IPCC guidelines for national greenhouse gas inventories: Chapter 11: N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. Prepared by the National Greenhouse Gas Inventories Programme, Hayama, Japan. https://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. Accessed 17 Feb 2021

IPCC (2019) 2019 Refinement to the 2006 IPCC guidelines for national greenhouse gas inventories: Chapter 11: N_2O emissions from managed soils, and CO_2 emissions from lime and urea application, Switzerland. https://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. Accessed 17 Feb 2021

ISCC System GmbH (2021) ISCC EU 205 greenhouse gas emissions. https://www.iscc-system.org/wp-content/uploads/2021/07/ISCC_EU_205_Greenhouse-Gas-Emissions-v4.0.pdf. Accessed 6 March 2022

Lange N, Majer S, Meisel K, Oehmichen K (2020) Assessed, assessing and to be assessed: past, present and future of LCA at the DBFZ. In: DBFZ Jahrestagung 2020: Bioenergie zwischen Klimapaket und Bioökonomiestrategie. Leipzig, 16./17. September 2020. DBFZ, Leipzig, pp 290–291

Long A, Bose A, O'Shea R, Monaghan R, Murphy JD (2021) Implications of European Union recast renewable energy directive sustainability criteria for renewable heat and transport: case study of willow biomethane in Ireland. Renew Sustain Energy Rev 150:111461. https://doi.org/10.1016/j.rser.2021.111461

Manninen K, Koskela S, Nuppunen A, Sorvari J, Nevalainen O, Siitonen S (2013) The applicability of the renewable energy directive calculation to assess the sustainability of biogas production. Energy Policy 56:549–557

McManus MC, Taylor CM (2015) The changing nature of life cycle assessment. Biomass Bioenergy 82:13–26. https://doi.org/10.1016/j.biombioe.2015.04.024

Naumann K, Müller-Langer F, Meisel K, Majer S, Schröder J, Schmieder U (2021) Further development of the German greenhouse gas reduction quota: Background paper, Leipzig. https://www.dbfz.de/en/press-media-library/more-publishments/statements-studies. Accessed 10 March 2022

Rana R, Ingrao C, Lombardi M, Tricase C (2016) Greenhouse gas emissions of an agro-biogas energy system: estimation under the renewable energy directive. Sci Total Environ 550:1182–1195. https://doi.org/10.1016/j.scitotenv.2015.10.164

Rasi S, Timonen K, Joensuu K, Regina K, Virkajärvi P, Heusala H, Tampio E, Luostarinen S (2020) Sustainability of vehicle fuel biomethane produced from grass silage in Finland. Sustainability 12:3994. https://doi.org/10.3390/su12103994

Rauh S (2010) Konkurrenz um Biomasse – Entwicklung eines Landnutzungsmodells zur Ableitung möglicher zukünftiger Entwicklungen in der Landwirtschaft hinsichtlich der Bereitstellung von Nahrungsmitteln und Energiebiomasse. Dissertation, Technische Universität München

Roßmann M, Stratmann M, Rötzer N, Schäfer P, Schmidt M (2019) Comparability of LCAs—review and discussion of the application purpose. In: Progress in life cycle assessment 2019, vol 2019, pp 213–225

Sala S, Amadei AM, Beylot A, Ardente F (2021) The evolution of life cycle assessment in European policies over three decades. Int J Life Cycle Assess 83:126. https://doi.org/10.1007/S11367-021-01893-2

Smyth BM, Murphy JD, O'Brien CM (2009) What is the energy balance of grass biomethane in Ireland and other temperate northern European climates? Renew Sustain Energy Rev 13:2349–2360. https://doi.org/10.1016/j.rser.2009.04.003

Timonen K, Sinkko T, Luostarinen S, Tampio E, Joensuu K (2019) LCA of anaerobic digestion: emission allocation for energy and digestate. J Clean Prod 235:1567–1579. https://doi.org/10.1016/j.jclepro.2019.06.085

Vogt R (2008) Basisdaten zu THG-Bilanzen für Biogas-Prozessketten und Erstellung neuer THG-Bilanzen: Kurzdokumentation. Im Auftrag der E.ON Ruhrgas AG, Altenessen, Heidelberg

Wietschel M et al. (2019) Klimabilanz, Kosten und Potenziale verschiedener Kraftstoffarten und Antriebssysteme für Pkw und Lkw, Endbericht, Karlsruhe

Whittaker C (2015) Life cycle assessment of biofuels in the European Renewable Energy Directive: a combination of approaches? GHG Measure Manage 4:124–138. https://doi.org/10.1080/20430779.2014.998442

Chapter 7
Proposing a Multi-level Assessment Framework for Social LCA and Its Contribution to the Sustainable Development Goals

Daniela Groiß-Fürtner, Claudia Mair-Bauernfeind, and Franziska Hesser

Abstract In the context of sustainable product development, Life Cycle Assessment (LCA) methods are used to gain knowledge about environmental hotspots and derive options for improvement. In light of international efforts to promote sustainable development, Social LCA (SLCA) is an emerging method to assess potential socio-economic impacts of products and services. Even when available data is limited in the early stages of materials, process, and product development, the implementation of SLCA benefits target-oriented research and development to support sustainable development. This article introduces a multi-level SCLA framework for accompanying innovation processes. The multi-level framework starts by prioritizing social aspects and proceeds as more and more data becomes available with generic and primary assessments and sets the results in context to the 17 Sustainable Development Goals (SDGs). The application of the multi-level SLCA is showcased via a bio-based value chain. The study aims to identify options for social risk reduction and consequently provide recommendations for decision-makers. The results show that options to increase social sustainability can be realized by reducing chemical and fertilizer use or fostering sustainability reporting. By mapping the SLCA results to the SDGs, it could be found that the bio-based value chain at hand mostly contributes to the SDG no. 8.

D. Groiß-Fürtner (✉)
Department of Economics and Social Sciences, Institute of Marketing and Innovation, University of Natural Resources and Life Sciences, Vienna, Feistmantelstraße 4, 1180 Vienna, Austria
e-mail: daniela.fuertner@boku.ac.at

C. Mair-Bauernfeind · F. Hesser
Wood K plus – Competence Center for Wood Composites and Wood Chemistry, Kompetenzzentrum Holz GmbH, Altenberger Straße 69, 4040 Linz, Austria

C. Mair-Bauernfeind
Institute of Systems Sciences, Innovation and Sustainability Research, University of Graz, Merangasse 18, 8010 Graz, Austria

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*,
Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_7

Keywords Social Life Cycle Assessment · Bio-based value chains · R&D · Sustainability · SDGs · Innovation process

7.1 Introduction

Since around 2005, there has been a growing interest in the social dimension of sustainability assessment, and from policymakers and industry to individuals from civil society, concern regarding the social impacts of production processes, products, or services has increased. Beginning with environmental impact assessment, the call for integration of social aspects in Life Cycle Assessment (LCA) arose later and together with economic aspects form the framework of Life Cycle Sustainability Assessment (McManus and Taylor 2015; Guinée et al. 2011). The first considerations to implement socio-economic aspects into LCA started in the early 1990s (Wu et al. 2014). Since then, formal guidelines have been developed, such as the "Guidelines for Social Life Cycle Assessment of Products" in (UNEP/SETAC 2009) in 2009. With an updated version in 2020, new and more advanced guidance was created for applying SLCA in practice (UNEP 2020). Social impacts are assessed by considering relevant social issues (impact categories), which are assigned to stakeholder groups and can be quantified and measured by a combination of various quantitative and qualitative indicators (UNEP/SETAC 2009; UNEP 2020). The stakeholder groups represent human beings who may be affected by the economic activities under study. Therefore, both positive and negative impacts on affected stakeholders throughout a product's life cycle can occur (UNEP/SETAC 2009).

However, assessing the social and socio-economic dimensions still faces some challenges: one issue is the availability of characterization models for impact pathways (Chhipi-Shrestha et al. 2015; Martínez-Blanco et al. 2014). The interdependencies of social and economic factors as well as the social cause and effect chains are complex and therefore difficult to quantify (Sureau et al. 2020; Spierling et al. 2018; UNEP 2021). For instance, land use change is mainly investigated from an environmental perspective (Rutz and Janssen 2014), although the rising establishment of bio-based value chains substituting for fossil-based products, propell expectations in the creation of wealth and job opportunities particularly for the rural population (European Commission 2012a; Global Bioeconomy Summit 2015). Another issue is the availability of adequate data on the different levels of assessment (i.e., site-specific, sectoral, and regional data), especially for ex-ante assessments during the innovation process (Hesser 2015; Mair-Bauernfeind et al. 2020a). The data available in the early stages of materials, process, or product development is inherently low and increases with the progression of the development process (Hetherington et al. 2014). Though, uncertainties are high in ex-ante assessments, it provides insights into areas of concerns already at low technology readiness levels (TRL) and guides further advancements when adaptations are still relatively easily possible (for integration of LCA in research and development, see Lettner and Hesser 2020).

Following the life cycle thinking approach, the Social Life Cycle Assessment (SLCA) methodology is being developed based on the standardized method of environmental LCA (ELCA) (see ISO 14040 series). The aim of both methods is to assess the potential social or respectively environmental impacts of products or services across their entire life cycle (i.e., from resource extraction to final disposal) (ISO 2006; UNEP 2020). As life cycles are related to complex value chains and are often geographically scattered, typically involving a range of stakeholders at every stage of the value chain, a multitude of social issues may be relevant for the assessment of social impacts. Also, the geographical, sector-specific, and even the company-related contexts are important in SLCAs (Sala et al. 2015; Dreyer et al. 2010; UNEP/SETAC 2009) not to mention cultural aspects and different sustainability issues in relation to geographical context (Sutterlüty et al. 2018). Consequently, certain methodological decisions (e.g., choice of stakeholder groups, subcategories or indicators) need to be adapted to sectoral and regional specifics (Siebert et al. 2018b; Mair-Bauernfeind et al. 2020b). For instance, Fürtner et al. (2021) prioritized social issues and indicators for bio-based value chains in Slovakia in a three-step multi-methodological approach, which resulted in a set of indicators specifically for the sectoral and regional context investigated.

For a sustainable development, the different dimensions of sustainability (economy, ecology, and society) must be considered, which needs an institutional framework to integrate the different dimensions (Zimmermann 2016). The 17 Sustainable Development Goals (SDGs) are a global plan to promote sustainable peace and prosperity as well as to protect our planet, launched by the United Nations, which came into force in 2016 under Agenda 2030 (UN General Assembly 2015). A broad set of targets and indicators for each SDG form the basis for evaluating the progress made towards achieving the SDGs operationalized also on a national scale. In the context of Life Cycle Sustainability Assessment (LCSA), the SDG indicators have already been considered as a support for indicator selection (Wulf et al. 2018) and were assigned to the impact categories (Maier et al. 2016) or were used to analyze the contribution to the SDGs (Herrera Almanza and Corona 2020). However, not every global goal and indicator may be relevant in region-specific assessments (Zeug et al. 2021) and when focusing on the social dimension, it is unclear how the SDG indicators will overlap with the prioritized impact categories. Moreover, as the SDGs represent targets on a macro level, they cannot be directly implemented into the assessment of products or services (micro level) and therefore need to be downscaled for application. Another issue is that up to the revised version of the "Methodological sheets for Subcategories in Social Life Cycle Assessment" published in 2021, the SLCA method had remained isolated from the SDGs (Pollok et al. 2021). Now the methodological sheets provide subcategories assigned to SDGs and—where applicable—to SDG targets (UNEP 2021). However, an implementation of SLCA results to the SDGs and the respective targets is missing. One reason for that might be that companies struggle with how to introduce, implement, and assess the contribution of their activities to the SDGs (Herrera Almanza and Corona 2020; Weidema et al. 2018). Nonetheless, companies have already started using the SDGs for their sustainability reporting, which plays a crucial role in achieving the SDGs, as companies will have an impact on the utilization of resources, stakeholder behavior, and innovation (Calabrese et al. 2021).

Considering this background, the aim of this study is twofold: First, a conceptual framework for a multi-level SLCA is introduced that allows accompanying the different research and development (R&D) phases with social sustainability assessment. A 2nd level assessment is showcased in this study where an SLCA is integrated into the development of a bio-based product system. Secondly, this article assigns the results of the SLCA to SDGs to depict the contribution of a product system to sustainable development. Hereby, positive impacts and the reduction of potential negative impacts to the SDGs are aimed at guiding sustainability efforts towards essential issues. Therefore, the SLCA results will be discussed based on the goals set by the SDGs. The study intends to address LCA practitioners as well as decision-makers (project managers) who are responsible for establishing bio-based product systems and developing innovations.

7.2 Methodological Approach

The multi-level SLCA and the application are showcased on the demonstration project D4EU (Dendromass4Europe), which aims to establish sustainable dendromass cultivation on marginal land in Western Slovakia valorizing the total biomass in four different bio-based value chains (D4EU 2022). Further information about the project can be found on the project's homepage (https://www.dendromass4europe.eu/). D4EU is funded by the Bio Based Industries Joint Undertaking under the European Union's Horizon 2020 research and innovation program. The technological development and demonstration in the project were accompanied by a set of investigations for sustainability assessment: see Perdomo et al. (2021) for LCA review on dendromass cultivation; Perdomo et al. (2022) on setting the dendromass production in context to planetary boundaries; Fürtner et al. (2021) on locating hotspots in social LCA; Ranacher et al. (2021) on willingness of farmers to engage in dendromass cultivation; Pichler et al. (submitted) on perceived fairness of dendromass production and its social license to operate; and Fürtner et al. (2022) on the costs and benefits of dendromass production.

The availability of data for products and processes at a low TRL (technology readiness level) is scarce but increases with advancing TRL. In this environment, LCA can make use of its iterative character and accompany the development progresses by co-developing the assessment models and providing preliminary assessment results and recommendations. Instead of waiting for the product to reach maturity and for accurate data to be made available, it is important to take precautions and assess the potential social impacts already at an early TRL so that unintended negative effects can be avoided (see also the European Commission's Responsible Research and Innovation framework) (European Commission 2012b). The precautions allow those managing a project to anticipate potential social hotspots already during the development process and provide the opportunity to counteract them in further development.

Fig. 7.1 Multi-level SLCA framework to accompany R&D projects/innovation processes along with their advancements

As this approach has already successfully been implemented in LCA studies (e.g., Lettner and Hesser 2020; Mair-Bauernfeind et al. 2020a), it should work for social sustainability assessments as well. Low data availability is often used as an excuse not to conduct LCA or another type of sustainability assessment. Therefore, we propose a multi-level SLCA framework from 1st level to 5th level (compare Fig. 7.1), which allows us to start assessments with open-ended process configurations and low or fragmented primary data availability.

1st level: "*Thematic Hotspotting*" presents the prioritization of social issues and indicators by indicator screening, stakeholder engagement, and risk mapping. This was done in a previous study by Fürtner et al. (2021), where a prioritization of relevant social aspects and indicators was reached by triangulating the results of several methods applied, including the following:

(a) Indicator screening (in guidelines, sustainability standards, and scientific articles);
(b) Stakeholder engagement (process experts, representatives of stakeholder groups, and affected stakeholders). This participatory approach allows to find social impacts that concern our stakeholders (Mathe 2014) and;
(c) Risk mapping (with the Social Hotspots Database Risk Mapping Tool available at http://www.socialhotspot.org/).

2nd level "*Impact Hotspotting*" includes country- and sector-specific secondary data assessment. The indicators selected in the previous level are quantified with national, regional, and statistical data. The procedure and results achieved at this level are discussed in the present article.

3rd level "*Organizational Level SLCA*" requires company-specific data, switching from a generic secondary data-based approach to a specific primary data-based approach assessing the social performance/impact of the organization.

4th level "*Process Level SLCA*" analyzes the stages of a value chain with corporate data of production processes. Cooperation of firms involved is necessary to obtain the required information. The results help identify process hotspots and to implement targeted measures, improving the social performance.

5th level "*Consequence-based SLCA*" integrates direct stakeholder consultation. This means that the social impact data are directly gathered from the different stakeholder groups that are potentially affected by the studied processes. Level five thus allows to assess directly perceived impacts, whereas at level 1–4, potential impacts can merely be investigated.

Depending on the goal of the assessment, it is not necessary to go through all levels but rather focus on individual levels of the multi-level SLCA framework that are useful for the project development. It is also strongly dependent on time and other resources to what level the SLCA can be applied. Therefore, one should avoid omitting an SLCA due to a lack of resources but instead concentrate on single levels that can be achieved.

The application of the multi-level framework is showcased by applying the 2nd level assessment on a bio-based value chain. In principle, the 2nd level assessment is based on the reference scale approach proposed in the SLCA guidelines by UNEP (2020), where the performance of a product system is assessed in relation to predefined performance reference levels (PRPs). The PRPs are intended to compare a local situation (described by the inventory data) with an national or international threshold (Parent et al. 2010; UNEP 2020). Performing the SLCA on a generic level as is proposed in the 2nd level assessment allows for identifying the most critical societal aspects (hotspots analysis) to find levers for supporting sustainable development in the value chain during R&D. The results are used to communicate potential pathways towards decision-makers to further increase the social sustainability through the value chain under study. In the discussion, the results are then linked to the global SDGs. This helps one to focus on generally accepted and globally relevant sustainability objectives and to contribute to the development towards the defined SDGs.

7.2.1 Goal and Scope

The aim of this study is to determine potential social impacts related to the product system introduced in Fig. 7.2, by conducting the 2nd level assessment of the introduced multi-level SLCA framework (Fig. 7.1).

In the 2nd level assessment, the focus is on the foreground system of the product system (Fig. 7.2), which includes those processes that can be directly shaped by the decisions made in the project (S0–S4). The system boundaries are, therefore, "cradle to gate" and include raw material production of dendromass (S0) in Slovakia to industrial production of four bio-based products: (S1) functionally adapted lightweight board in Slovakia; (S2) eco-fungicidal molded fiber parts in Poland; (S3) bark-enriched wood plastic composite profiles in the Czech Republic; and (S4) bark-enriched wood plastic composite granulate also in the Czech Republic. These production systems differ in their production processes, sector, and geographical scope. The

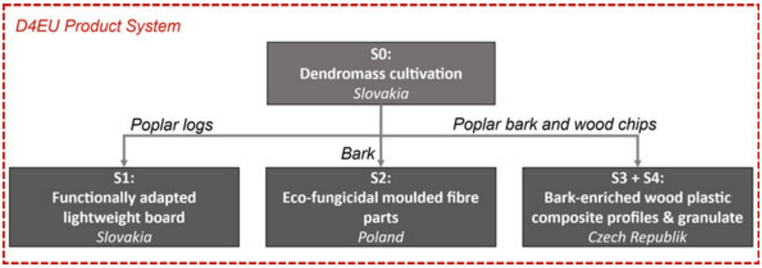


Fig. 7.2 Production system comprising the subsystems: S0 Dendromass cultivation, S1-S4 bio-based products production in different geographical contexts

UNEP (2020) guidelines propose six stakeholder groups to be investigated, in this study the central stakeholders that may be affected by the processes are as follows:

- Workers (field workers in SRC plantations and industrial workers),
- Local communities (neighboring to production systems) and
- Society (potentially affected groups of people in the region, federal state, state, etc.).

The other three stakeholder groups, namely value chain actors (people or organizations involved through a business relationship, e.g., suppliers), consumers, and children cannot be considered for this 2nd level assessment due to the challenging data situation. Likewise, the consumption and end-of-life phase is out of the scope for the 2nd level assessment.

The prioritized stakeholder groups are addressed in this 2nd level SLCA by assigning their impact categories and indicators, which have been prioritized in a pre-study (see Fürtner et al. 2021) and are used as a starting point in this study. The list of prioritized indicators and impact categories is shown in Table 7.1. The stakeholder category of value chain actors was also included in Fürtner et al. (2021) but not in this study because of data availability. The assessed impact categories and indicators have strong focus on the stakeholder groups worker and local community. This focus was also observed in other studies, like Martin et al. (2018), Siebert et al. (2018a), or Spierling et al. (2018).

7.2.2 Data Inventory

To "get a general feel for areas of social concerns in certain countries/or sectors" (Benoît-Norris et al. 2011, 687), the 2nd level assessment can be used for product systems in an early development stage. For such an assessment, modeling tools that require less accurate datasets can be used (Hesser 2015; Niero et al. 2014). In this study, the indicator results of the different production locations were compared, whereby different types of generic sectoral and country-specific data were collected.

Table 7.1 Social impact categories and indicators considered for the 2nd level assessment (based on Fürtner et al. 2021)

Stakeholder group	Impact categories	Indicators	Units	Measurement description
WORKERS	Workers health and safety	1. Occupational accident rate in Slovakia 2. Number of occupational (fatal) accidents 3. Sick-leave days per year 4. Exposure to agrochemicals	% nb nb Qual	2. Number of (fatal) accidents per year, per employee 3. Number of sick-leave days per year, per employee
	Equal opportunities	1. Country/region gender index ranking 2. Presence of formal policies on equal opportunities 3. Rate of female workers 4. Rate of workers from regional minorities	Index Yes/no % %	2. Description of potential discrimination practices
	Fair salary	1. Average Slovakian living wage (month) 2. Average payment per month, per full-time employee 3. Payment according to Slovakian living wage	€ € yes/no	3. Are all employees paid at least according to the local minimum wage?
	Working conditions	1. Job satisfaction	Index	Job satisfaction index
	Working hours	1. Contractual working hours 2. Effective working hours (average) 3. Effective used holidays 4. Overtime compensation	Hrs Hrs Days Qual	2. Hours of work per employee/day 3. Hours of consumed holidays per employee/ year
	Child labor	1. Percentage of children working by country and sector 2. Absence of working children under the legal age	% Yes/no	1. Description of child labor potential 2. Stating names, birth dates of all workers

(continued)

Table 7.1 (continued)

Stakeholder group	Impact categories	Indicators	Units	Measurement description
	Forced labor	1. Evidence of forced labor in the production processes 2. Workers voluntarily agree upon employment terms	Yes/no Yes/no	2. Description of working conditions contractually regulated
LOCAL COMMUNITY	Local employment	1. Unemployment statistics for Slovakia/Region 2. Percentage of workforce hired locally 3. Number of local full time equivalent created jobs	% % nb	
	Safe and healthy living conditions	1. Pollution levels by country 2. Management effort to minimize use of hazardous substances 3. Food security	% Qual Qual	3. Changes in national/local food prices
	Access to material resources	1. Changes in land ownership 2. Infrastructure for community access developed	Yes/no Qual	
	Community engagement	1. Number and quality of meetings with community stakeholders	nb./qual	1. Description of community engagement activities
	Cultural heritage	1. Strength of policies in place to protect cultural heritage 2. Landscape identity	Yes/no Qual	2. Visual attractiveness and continuity of appreciated landscape heritage
	Respect of indigenous/local communities rights	1. Prevalence of racial discrimination 2. Local land rights conflicts/land claims 3. Annual meetings held with community members	Yes/no Yes/no nb	2. Description of conflicts, land tenure structures, etc

(continued)

Table 7.1 (continued)

Stakeholder group	Impact categories	Indicators	Units	Measurement description
	Regional value creation	1. Regional Value Added 2. Regional investment, per unit input 3. Spatial proximity of investments	€ € %/qual	
	Contribution to (regional) economic development*	1. Economic situation of country/region 2. Contribution to economic progress 3. Contribution to household/farm income	€/qual €/qual €/day	1. GDP, economic growth, unemployment rates, wage level, etc 2. Revenues, paid wages, R&D costs, etc
SOCIETY	Public commitments to sustainability standards	1. Existence of public sustainability reporting 2. Publicly available documents on agreements to sustainability issues	Yes/no Yes/no	
	Corruption	1. Risk of corruption in Slovakia/the region 2. Commitment to prevent corruption 3. Anti-corruption program carried out	Index Yes/no Yes/no	
	Technology development	1. Research and development costs spent 2. Partnerships in R&D	€ Yes/no	1. On organizational, sectoral, or project level
	Respect of human rights	1. Slovakian Human Rights Index	Index	1. CIRI Human Rights Data Project

* Contribution to economic development can be considered either from the perspective of a whole society or from the perspective of a smaller local community

Secondary data was used to relate the potential social risk associated with the country or sector of production. The secondary data utilized for this study were based on various national- and sector-specific statistics and literature gathered through online research (from databases like WHO, OECD, or ILO). Information needed for the qualitative description of certain indicators was also collected through online research. Accordingly, primary data, which is company-specific information in this case was collected for those indicators, where a generic approach does not make sense. The use of a functional unit (FU) is not necessary for this 2nd level assessment because an impact on stakeholders does not depend on production volumes, but on the principles of decision-makers in different countries, sectors, and companies (UNEP/SETAC 2009; Iofrida et al. 2018; Martínez-Blanco et al. 2014).

7.2.3 Impact Assessment

The Life Cycle Impact Assessment (LCIA) in SLCA is applying characterization methods to link the inventory data to the respective impact categories and to calculate the potential impact of the system with the help of indicators (UNEP 2020). These results are normalized in relation to reference information to bring them on a common scale for comparability (Ibáñez-Forés et al. 2019; Yıldız-Geyhan et al. 2019). This approach allows to compare the local situation (inventory data) with an international set of thresholds (e.g., EU-average of the respective data) (Parent et al. 2010). Since the data availability for the assessment of the indicators is quite inconsistent, individual reference values (e.g., EU average, national or organizational target values or, best practice targets, etc.) were used to assess each of the indicators as accurately as possible. The social risk potential is calculated following the method of Zira et al. (2020) with the Eqs. (7.1) and (7.2), introduced below.

$$SR = 1 - EXP\left(LN(0.5) \times \frac{IND}{REF}\right) \tag{7.1}$$

when a higher value than the reference point reflects a more negative impact, and

$$SR = EXP\left(LN(0.5) \times \frac{IND}{REF}\right). \tag{7.2}$$

when a lower value than the reference point reflects a more negative impact.

$\text{SR} = \text{Social Risk Potential};\ \text{IND} = \text{the inventory indicator};\ \text{REF} = \text{the reference point}.$

7.2.4 Interpretation and Discussion of the Results Based on the SDGs

The interpretation phase of SLCA facilitates identifying significant risks as well as drawing conclusions and offering recommendations on the results as well as checking for completeness, consistency, and limitations of the assessment (UNEP 2020). Considering that the SLCA is intended to provide guidance on improving the social sustainability of the evolving value chain, emphasize is given to translate the results into recommendations that support decision-making. In order to pursue a global strategy of sustainable development, the results are interpreted in terms of the SDGs. In this regard, the frequently used technique to identify direct and indirect impacts of the evolving value chain on the achievement of the SDGs is used (Eberle et al. 2022). Whereas, a direct impact is given, when the value chain under study has an influence on the fulfillment of SDGs by its own production processes and activities. Indirect influence originates from outsourced processes in up- and downstream activities of the value chain, which are managed by external companies or parties. The recently published methodological sheets for SLCA is an updated version where the SDGs are put in relation to the respective impact categories (UNEP 2021). This document is also used as a guideline for assigning project impacts to SDGs.

For this study, it is not the goal to follow a strategy of completeness, i.e., that each SDG or SDG-indicator needs to be covered through the assessment also because not every global indicator inevitably has to be relevant for regional considerations either (Zeug et al. 2021). The approach is intended to provide guidance on pursuing proper or generally accepted goals to avoid focusing on minor social issues. To avoid judging the impact of the project on the achievement of the SDGs solely based on predetermined associations from literature and preliminary studies, project-specific impacts were elicited in a first step. Following, the relationship between global SDGs and project-specific impact categories was linked (compare Fig. 7.6).

7.3 Results of the 2nd Level SLCA

The social risk potential indicates the likelihood in how far the assessed category has a higher or a lower risk than the related PRP. The corresponding social impacts for the stakeholder groups "Workers," "Local Communities," and "Society" is presented in the Sects. 7.3.1–7.3.3. The social impact categories and indicators from Fürtner et al. (2021) were directly applied if data were available for assessing the respective indicators. The results for the product systems (S0-S4) of the bio-based value chain as well as the respective reference levels (PRP level) for all indicators are shown in Table 7.2. The numbers indicate the normalized results in relation to the PRPs and are illustrated in Figs. 7.3, 7.4 and 7.5. The results can be interpreted as follows: 0–0.35 "above compliance/better social performance"; 0.35–0.7 "around compliance/rather satisfying social performance" and 0.7–1 "non-compliance/relatively poor social performance" compared to the PRP level.

Some social aspects cannot be evaluated with quantitative data. In such cases, qualitative data is used to describe the situation even if it is not possible to include that aspect into the SLCA rating system (e.g., food security). The following results refer to the "2nd level" assessment of the introduced multi-level framework. These findings result in a country- and sector-specific hotspotting of potential social impacts.

Table 7.2 Results of the 2nd level assessment for the systems S0–S4 and their respective PRP level

Performance Indicator	S0	S1	S2	S3 + S4	PRP level
Age-standardized all-cause disability-adjusted life year (DALY) rates attributable to occupational risks (per 100,000)	0.442	0.442	0.539	0.560	EU-27
Fatal accidents at work, by sector per 100,000 inhabitants	0.922	0.498	0.477	0.401	EU-27
Non-fatal accidents at work, by sector per 100,000 inhabitants	0.243	0.261	0.268	0.391	EU-27
Percentage of unemployed people with basic education (% of total labor force)	0.771	0.771	0.317	0.408	EU
Percentage of unemployed people with advanced education (% of total labor force)	0.332	0.332	0.203	0.152	EU
Percentage of unemployed people from the Roma Communities in Slovakia	0.997	0.997	–	0.979	National
Global Gender Gap Index—Economic Participation and Opportunity	0.563	0.563	0.540	0.566	EU-27
Risk of sector average wage being lower than country's average wage	0.518	0.495	0.514	0.509	National
Risk of sector average wage being lower than country's living wage	0.287	0.264	0.135	0.208	National
Risk of sector average wage being lower than country's minimum wage	0.215	0.194	0.266	0.168	National
Fair wage potential (Neugebauer et al. 2017)	0.202	0.159	0.313	0.135	National
Global Employee Engagement Index	0.515	0.515	0.515	0.500	EU
Average usual working hours per country	0.529	0.529	0.532	0.529	EU-27
Realization of Children's Rights Index Ranking	0.546	0.546	0.537	0.534	Best practice
Realization of Children's Rights Index Ranking	0.502	0.502	0.493	0.490	EU-27
Country's risk of forced labor used to produce commodity—Global Slavery Index Ranking	0.522	0.522	0.538	0.512	EU-27
Percentage of forced labor by country	0.484	0.484	0.531	0.477	EU-27
Government response rating of legal, policy, and programmatic actions to modern slavery	0.525	0.525	0.525	0.525	EU-27
Vulnerability to modern slavery by country	0.640	0.640	0.601	0.512	EU-27
Freedom of association and collective bargaining—"ITUC Global Rights Index"	0.500	0.500	0.875	0.750	Best practice
Risk of unemployment (% of total labor force)	0.492	0.492	0.298	0.254	EU
Human development index (HDI)	0.516	0.516	0.508	0.500	EU-27
DALY rates from all causes (per 100,000)	0.541	0.541	0.535	0.508	EU-27

(continued)

Table 7.2 (continued)

Performance Indicator	S0	S1	S2	S3 + S4	PRP level
Population-weighted mean levels of fine particulate matter smaller than 2.5 microns (PM 2.5)	0.705	0.705	0.765	0.672	WHO threshold
ENAR recorded incidents of racial motivated crime, per 100,000 inhabitants	0.181	0.181	0.523	0.158	EU-27
EU Regional Competitiveness Index (RCI)	0.302	0.302	0.4516	0.530	National
EU Regional Competitiveness Index (RCI)	0.416	0.416	0.623	0.530	EU-27
Annual growth in GDP per country	0.680	0.680	0.481	0.655	EU
Sector contribution to GDP per country	0.463	0.425	0.423	0.368	EU-27
Company publishes public sustainability reporting	0.287	0.287	0.871	0.871	Best practice
Company makes documents on agreement to sustainability issues publicly available	0.287	0.287	0.871	0.871	Best practice
Corruption Perceptions Index (CPI)	0.587	0.587	0.544	0.556	EU-27
Global Corruption Index (GCI)	0.627	0.627	0.561	0.535	EU-27
Company fosters partnerships in R&D	0.435	0.435	0.435	0.435	Best practice
Human Rights Score	0.580	0.580	0.578	0.406	EU-27
Human Rights Violations	0.551	0.551	0.526	0.473	EU-27

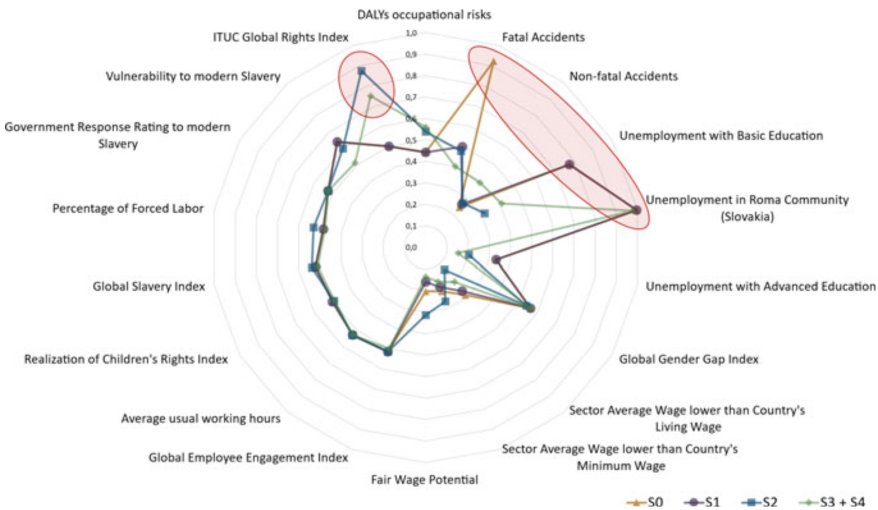


Fig. 7.3 Comparison of SLCA results among S0–S4 for the stakeholder group "Workers." Results in the respective range indicate: 0–0.35 "above compliance/better social performance" | 0.35–0.7 "around compliance/rather satisfying social performance" | 0.7–1 "non-compliance/relatively poor social performance" compared to the reference

7.3.1 Social Risk Concerning the Stakeholder Group "Worker"

The stakeholder group "Workers" is assessed with 20 indicators associated to eight impact categories (cf. Table 7.2). Five production systems (S0–S4) are compared to each other, whereas S3 and S4 are combined because they are produced within the same company located in the Czech Republic. The results are shown in Fig. 7.3, where 13 out of the 20 indicators for S0 (dendromass production system) have a social risk potential equal or higher than 0.5, which means that the situation is worse than the PRPs" situation and special attention should be paid to these aspects. The results are similar for S1 (bio-based product manufactured in Slovakia), S2 (bio-based product manufactured in Poland) as well as S3 and S4 (bark-enriched wood plastic composite profiles and bark-enriched wood plastic composite granulate) where 11 out of the 20 indicators yield in a value equal or higher than 0.5.

A major difference between the systems could be identified for the indicator "**fatal accidents at work**" In this area, agricultural and forestry activities impose far more risks than industrial activities. Fatal accidents in SRC production are one of the main hotspots that could be identified for dendromass production and should therefore be given special attention. Health and safety at work is a highly important topic to achieve sustainable development (Benoît-Norris et al. 2013). However, it must be mentioned that this assessment is based on statistical numbers also including timber logged in forests, which is associated with an inherently high risk. SRC specific statistics cannot be found; therefore, it is quite unsure to what extent the risk applies to dendromass production in SRC plantations. Discussions with SRC managers, harvesting experts, and planting companies, for example, indicated that the risk of accidents is significantly lower in SRC plantations than in forests due to a high degree of mechanization and controlled environment. No severe incidents were reported by them.

A rather high-risk potential could be identified regarding "**unemployed people with basic education**" for systems situated in Slovakia (S0, S1), and, for "**unemployed people from Roma communities**" for systems situated in Slovakia and Czech Republic (S0, S1 and S3, S4), which implies inequalities due to socio-demographic attributes. In this context, emphasis can be given to communicate possibilities of employment, as jobs are created with the establishment of an SRC-based value chain that could appeal to the affected groups of people. Another high-risk potential is connected with the "**global rights index**" for the systems S2–S4. This index documents violations of internationally recognized labor rights by governments and employees. The high rating for Poland and the Czech Republic indicates, that violations of collective labor rights are regularly reported (ITUC 2020).

A range of indicators were identified to be compliant with the reference level, which can be explained by relatively similar conditions among EU countries. Regarding the issue of a fair wage, the results show a relatively low risk potential, especially for the industrial activities referring to S1–S4.

7.3.2 Social Risk Concerning the Stakeholder Group "Local Community"

The stakeholder group "Local Communities" is assessed with eight indicators in four impact categories. The results shown in Fig. 7.4 indicate that the majority of assessed indicators result in a low or medium risk potential. A higher risk potential for "**incidents of racially motivated crime**" was detected for S2, which concerns processes located in Poland, indicating that human rights of marginalized groups are threatened by cultural mainstream. Organizations should emphasize respect to local cultural heritage as well as secure individuals" rights to preserve their cultural heritage (Benoît-Norris et al. 2013). In contrast, this threat is considered to be much lower in Slovakia and the Czech Republic, however, this does not imply that this aspect can be neglected by the companies responsible for S0, S1, S3 and S4. A major hotspot of social risks for local communities occurs in areas of high "**levels of fine particulate matter (PM 2.5)**," affecting the health of communities regarding all five systems under study. This issue should be addressed by all participating companies putting sustainable managing practices as a priority and focusing on the reduction of emissions, which contribute to the increase of PM 2.5 (e.g., caring about the reduction of vehicle use, transportation distances or emissions through incineration). Another high-risk potential could be found in the systems located in Slovakia and the Czech Republic regarding "**annual growth in GDP**," which simply means that those countries have a lower GDP growth than the EU average. According to the methodological sheets of UNEP (2021), organizations contribute to economic development by not only generating revenue but also by creating jobs, education, and training, they make investments and forward research. On a national level, this means that countries with lower GDP growth have a higher risk of contributing less to the abovementioned aspects. Note that the UNEP SETAC guidelines rate low GDP as social risk, which makes clear that the economic growth paradigm is supported.

"**Food security**" within the impact category "**safe and healthy living conditions**" cannot be described in a quantitative way as no respective data is available. However, the topic is highly discussed in the context of plantation establishment. One indicator that may illustrate this aspect is the "**Global Hunger Index**." Slovakia reached a score of 6.4 for the year 2020, which shows low severity for the risk of hunger. The "Global Hunger Index" measures and tracks the hunger situation at a global, regional, and national level from <9.9, indicating low risk to >50, indicating a severe situation (Grebmer et al. 2020). Slovakia is ranked on the 27th position out of 107 countries covered in the index. It was not possible to include the index into the generic SLCA, as reference levels were missing. For most of the EU-27 countries, the index has no value included (probably as it is no issue in the Central European context). However, it must be noted that among the included European countries (Estonia, Latvia, Lithuania, Romania, Croatia and, Bulgaria), the score for Slovakia shows the worst situation. Another indicator on that topic, which could not be included into the SLCA because of a missing reference is the number of **undernourished people in the country**. Undernourishment is measured as a caloric intake that is insufficient to meet the

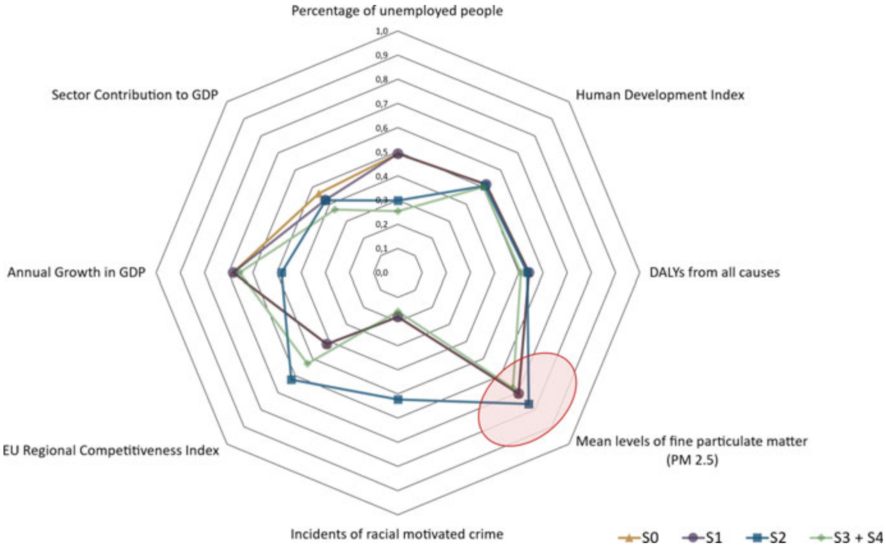


Fig. 7.4 Comparison of SLCA results among S0–S4 for the stakeholder group "Local Community." Results in the respective range indicate: 0–0.35 "above compliance/better social performance" | 0.35–0.7 "around compliance/rather satisfying social performance" | 0.7–1 "non-compliance/relatively poor social performance" compared to the reference

minimum energy requirements necessary for a given individual (Ritchie 2022). The results for this indicator are not showing rates below 2.5%, which indicates that the situation is not problematic at all. In that context, it must be mentioned that Slovakia and Bulgaria are the only two countries among EU-27 countries being included in this indicator. It was stated that in 2017, 200,000 people in Slovakia were malnourished. This risk increases the need to emphasize the selection of land for SRC plantations, using marginal land not suitable for food production to not compete with food and fodder production. This situation underlines the importance of dealing with issues in SLCA, which cannot be covered by the straight-forward quantitative assessment method but play a critical role.

7.3.3 Social Risk Concerning the Stakeholder Group "Society"

The stakeholder group "Society" is assessed with seven indicators in four impact categories. The results are shown in Fig. 7.5. The impact category "**commitment to sustainability issues**" is the only category of the 2nd level assessment, which was assessed with organization-specific information. This information could be obtained by online research, as a purely generic level would not make sense in this context. The "**existence of public sustainability reporting**" (corporate non-financial reporting

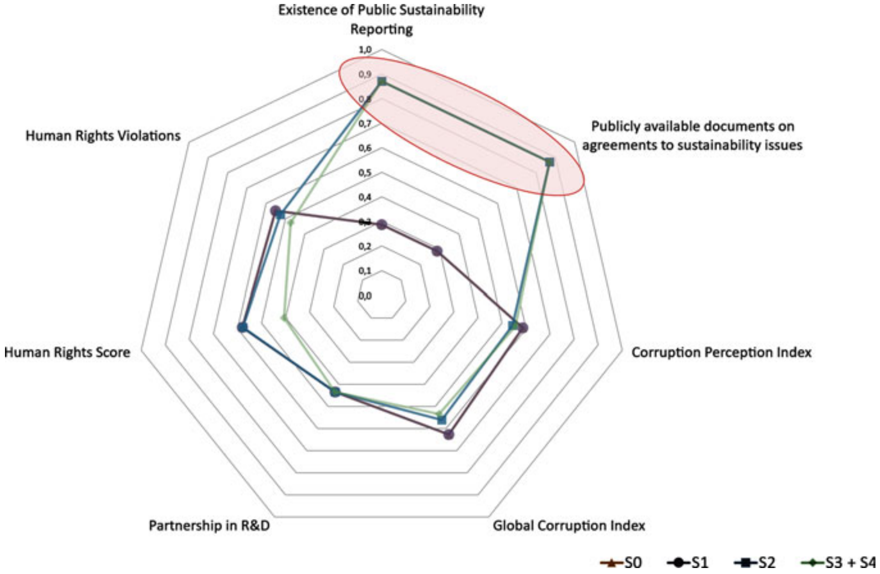


Fig. 7.5 Comparison of SLCA results among S0–S4 for the stakeholder group "Society." Results in the respective range indicate: 0–0.35 "above compliance/better social performance" | 0.35–0.7 "near compliance" | 0.7–1 "non-compliance/relatively poor social performance" compared to the reference

on environmental and social issues) and the "**availability of public documents on agreements to sustainability issues**" were assessed by publicly available sources. Only the organizations responsible for S0 and S1 publish documents on their sustainability performance; therefore, their score is relatively good in comparison to the other systems. This could be taken as an opportunity to strive for transparent documentation on the sustainability situation within the respective companies.

Not a severe, but a medium risk potential was observed for the indicators "**human rights violations**," "**human rights scores**," "**global corruption index**," and "**corruption perception index**." The **human rights scores** measure the degree to which governments protect and respect human rights, whereas the **human rights violations** indicator is an index including press freedom, civil liberties, political freedom, human trafficking, political prisoners, incarceration, religious persecution, torture, and executions (Herre and Roser 2016). The **corruption perception index** ranks countries according to the probability of corruption within the public sector of a country (Transparency International 2021). In comparison, the **global corruption index** includes the ratification status of key conventions, corruption perception, corruption experience, country characteristics, membership to FATF (Financial Action Task Force) and/or related bodies, money laundering and financing terrorism (Global Risk Profile 2020). Although, these issues just demonstrate a medium risk potential, they still should be taken seriously by all acting companies within the value chain under establishment, as the risk is still higher than the EU-average, although

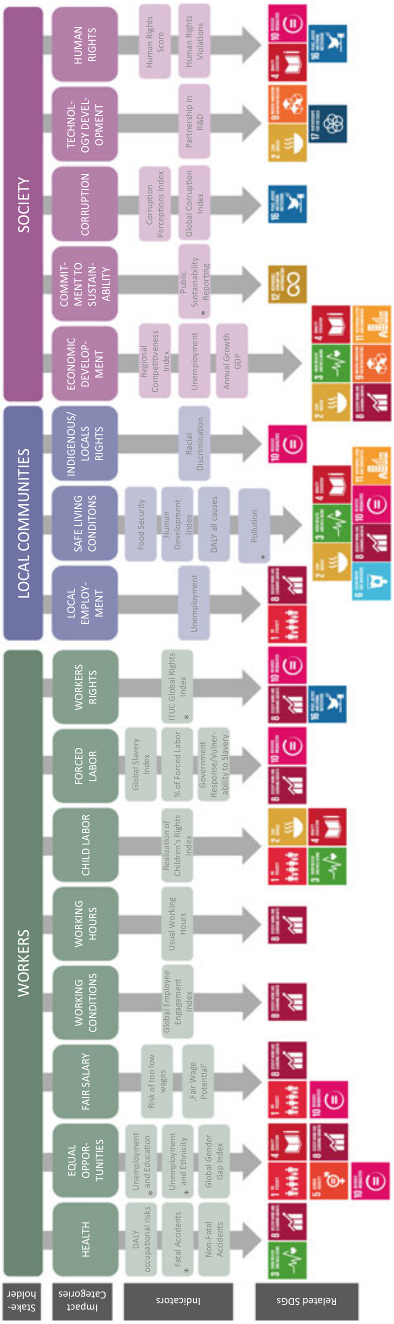


Fig. 7.6 Connecting the prioritized SLCA impact categories with the 17 SDGs. * marks the indicators with the highest risk rating

the assessment does not show a severe risk. Companies need to adopt a binding code of conduct and/or implement systems that prevent the company from getting engaged in corrupt activities (Benoît-Norris et al. 2013).

7.4 Interpretation of SLCA Based on the SDGs Framework

The 2nd level SLCA results for the bio-based production system under study shows a range of indicators with high-risk potential for the considered stakeholder groups. In this section, the social issues with the highest risk-potential will be discussed based on their respective relevance to the SDGs. Thus, the contribution of the bio-based industry to meet the goals will be discussed. The mapping of the impact categories and indicators prioritized for the bio-based value chain under study to SDGs was carried out based on UNEP (2021) and is shown in Fig. 7.6. The SDGs are global goals on a macroeconomic level, thus, comprising governmental objectives. This is probably the greatest challenge of integrating the SDGs into sustainability assessments (Wulf et al. 2018; Herrera Almanza and Corona 2020). Although there is a lack of research on SDGs-based sustainability assessment (Eberle et al. 2022), there are published approaches to integrate the SDGs into the assessment of products (e.g., Eberle et al. 2022) or the prioritization of impact categories and indicators for sustainability assessments (e.g., Wulf et al. 2018). To avoid downscaling and associated inaccuracies, the 2nd level assessment method deals with the SDGs in the interpretation phase. As the 2nd level assessment is based on country- and sector-specific data, the results are located at a similar level as the SDGs. In the following, the indicators with the highest risk rating are discussed in relation to the SGDs (marked with an * in Fig. 7.6) and per social impact category (see Table 7.1).

Workers health and safety

A relatively high-risk potential is associated to "fatal accidents" within the dendromass production system, whereas the industrial production systems (S1–S4) are associated with a lower risk. This is attributable to the different sectors, as the agricultural sector (including farming, forestry, and fishery) shows a considerably higher risk potential than the industry sector. Agriculture and forestry are one of the most dangerous sectors for workers in Europe (Jones et al. 2020). However, it is anticipated that digitalization and new technologies will help navigate the sector towards work practices with higher safety standards (Jones et al. 2020). Established harvesting options are fully mechanized and require relatively low manpower for handling the machineries that may lower the risk of occupational accidents. The prevention of work incidents contributes to ***SDG 8: promote sustained, inclusive, and sustainable economic growth, full and productive employment, and decent work for all*** as well as ***SDG 3*****:** ***ensure healthy lives and promote well-being for all***. The reduction of occupational injuries directly contributes to target 8.8., which aims to promote safe and secure working environments for all workers. The single targets of SDG 3 do not include the prevention of accidents at work or similar. This could be explained by the fact that the SDGs focus on a broader perspective concerning political organizational units. However, target 3.9, which deals with the reduction of deaths and illnesses from hazardous chemicals as well as air, water, and soil pollution and contamination, may be influenced by the bio-based production system although low amounts of fertilizers and pesticides are needed (Ranacher et al. 2021). These aspects were not included in the SLCA, as they are assessed by ELCA (Perdomo et al. 2022). Nonetheless, a reduction of chemical and fertilizer use in plantations is also contributing to people's health.

Equal opportunities and the freedom of association and collective bargaining

Even though many workers are not needed to manage the SRC plantations (Ranacher et al. 2021), the establishment of a new bio-based value chain in rural areas is seen as an opportunity to strengthen rural areas and create job opportunities. The results of the SLCA show that unemployment rates within the Roma community in Slovakia and the Czech Republic are relatively high. People who have only basic education are also affected by higher unemployment in Slovakia, whereas the situation is much better for the system S2, S3 and S4 located in Poland and the Czech Republic. Therefore, the creation of jobs, especially for lower qualified people and for people from different cultural backgrounds, is of high importance. However, the job market in Slovakia as well as in other countries is being negatively impacted by the COVID-19 pandemic, and people from other industries (e.g., accommodation and food services) are available for employment (Svabova et al. 2020).

For the system S2 (Poland) and S3, S4 (Czech Republic), the "ITUC Global Rights Index" indicator shows a relatively high-risk potential, which indicates that companies should make sure that they are compliant with freedom of association and collective bargaining standards. In contrast, a relatively low risk could be assessed for all systems (S0–S4) regarding the wage associated indicators. These issues contribute mainly to ***SDG 8:promote sustained, inclusive and sustainable economic growth, full and productive employment, and decent work for all***. A contribution can be made for a range of targets within SDG 8 when the established value chain promotes sustainable economic growth and higher levels of productivity (8.1), diversification and innovation (8.2), job creation (8.3), resource efficiency (8.4) and, productive employment for all (8.5 and 8.6). Even though, the 2nd level SLCA results show severe risk potentials in the inclusion of socio-economically disadvantaged groups in the job market. It should be seen as an opportunity to create jobs for them through the establishment of a diversified and innovative value chain. Resource efficiency is again a matter of ELCA; however, it is an important goal to encourage ecological economic growth. All segments of the value chain under establishment should follow the rules to save resources (e.g., through a cascading use of the fully harvested dendromass). SDG 8 is seen as the goal with the highest potential to be positively affected by a sustainable bioeconomy (Allen et al. 2020).

Safe and healthy living conditions and the contribution to economic development

Concerning the stakeholder group local communities, most of the indicators within this group show a relatively low or equal risk potential compared to the reference levels. The highest risk-potential is shown for the fine particulate matter (PM 2.5) situation in Slovakia, Poland, and the Czech Republic, what concerns all production systems under study. PM 2.5 is a common measure for air pollution, which is described by the WHO (World Health Organization) as one of the major risks for human health in all countries of the world (WHO 2021). Air pollution is estimated to be responsible for 4.2 million premature death worldwide, causing heart, lung, and respiratory diseases (WHO 2021). Direct measurements of PM 2.5 emissions through the establishment of the bio-based value chain may be subject of ELCA calculations. However, the project management can contribute to the reduction of PM 2.5 levels by promoting the reduction of transportation distances or the reduction of incineration of wood through avoiding waste in the production processes. These effects can be mainly attributed to ***SDG 11: making cities and human settlements inclusive, safe, resilient, and sustainable*** as well as ***SDG 3***. Especially, the reduction of environmental impacts in cities with a focus on better air quality is topic of SDG 11 (target 11.6). The reduction of human death and illness incidents due to pollution is closely linked to this—and is pursued in SDG 3 (target 3.9). In Slovakia the score of annual mean concentration of PM 2.5 is moderately improving, however, is insufficient to attain the goal (Sachs et al. 2022). The situation in Poland is even rated a little worse (Sachs et al. 2022), which associates a potential to support the fulfillment of SDG 11 and 3 with the reduction of air pollution. Another aspect in this impact category is food security. Although this aspect could not be measured in quantitative terms and was not included in the assessment because of a missing reference level, it is highly discussed in the context of plantation establishment. All countries involved in the bio-based value chain show a low severity for risk in the global hunger index or undernourishment. However, among the included European countries, Slovakia performs worst and in 2017, 200,000 people in Slovakia were malnourished (Ritchie 2022). The aspects of food security contribute to **SDG 2: end hunger, achieve food security and improved nutrition, and promote sustainable agriculture**, especially target 2.2, which aims to end all forms of malnutrition. This situation underlines the importance of dealing with issues in SLCA, which cannot be covered by the straightforward quantitative assessment method but play a critical role.

Public commitments to sustainability standards

The highest risk potential, within the indicators concerning the stakeholder group society, is the non-existence of publicly available sustainability reporting and agreements on sustainability issues. The underlying cause of this result is that reporting, and agreements are missing for the systems S2–S4. However, the results are quite severe, as the indicators only allow a “yes” or “no” answer. These two aspects regarding commitment to sustainability can be assigned to ***SDG 12: ensure sustainable consumption and production patterns***. Target 12.6 directly deals with adopting sustainable practices and integrating sustainable information into regular reporting. The reason for promoting sustainable production with sustainability reporting may be that sustainable efforts can be pushed forward through setting binding targets and KPIs. Following this path, the targets 12.2—promoting the efficient use of natural resources, 12.4—dealing with a sound management of chemicals and waste along the value chain, and 12.5—the reduction of waste through prevention, reduction, recycling, and reuse measures can be contributed to by setting targets and KPIs in the sustainability reporting.

Focusing on social sustainability in this study has shown, that mainly SDGs 3, 8, 11, and 12 can be influenced by following sustainable pathways. Compared with the study of Allen et al. (2020), our result is more restrictive than their prioritized SDGs for a bioeconomy, focusing on SDGs 2, 6, 9, 12, 13, 14, and 15, which may be due to the fact that their study focused on all dimensions of sustainability. Heimann (2019) found that a bioeconomy can have a negative impact on SDG 1 ("No Poverty") due to an increase of land demand having the same effects as industrial agriculture (land grabbing, displacements, etc.) as well as positive impacts on SDG 1 (income for farmers, higher value-added industry). This aspect could not be supported with our study or is missing within the assessed impact categories. However, Heimann (2019) found that there is the opportunity of a positive effect through a bioeconomy on SDG 8, which can be confirmed with our results.

7.5 Conclusions

The following conclusions and recommendations are derived to contribute to sustainable development of the bio-based production system/demonstration project under study:

- Promote less/no chemical and fertilizer use in plantations and fully mechanized harvesting and planting technologies to support healthy lives and well-being for all stakeholder groups (SDG 3 and SDG 8).
- Promote diversification (farms) and innovative working practices to create job opportunities especially for disadvantaged groups on the job market (SDG 8).
- Increase resource efficiency within the establishment of the bio-based value chain, to encourage the decoupling of economic growth from environmental degradation (SDG 8).
- Set measures to reduce transportation distances and incineration of wood to reduce air pollution and promote a better air quality and human health (SDG 11 and 3).
- Request sustainability reporting from companies acting within the value chain under establishment to push forward sustainable production patterns by fixed targets and KPIs (SDG 12).

Given the limited data availability, cooperation of firms, or normative underpinnings in methodologies (e.g., necessity of economic growth), the study at hand shows that already with an early stage SLCA (2nd level assessment), it is possible to generate and derive actionable knowledge to contribute, identify, and mitigate social risks. In conclusion, the knowledge of driving factors gained through the 2nd level assessment, and its results mapped to the SDGs will provide a strategy leading to increased social sustainability for decision-makers in the development of the project.

Acknowledgements This work received funding by the Bio Based Industries Joint Undertaking under the European Union's Horizon 2020 research and innovation programme under grant agreement number 745874 and the Austria Research Promotion Agency (FFG) under the COMET program grant number 865905.

References

Allen B, Nanni S, Bowyer C, Kettunen M, Giadrossi A (2020) Assessing contribution towards the SDGs: Guidance for evaluating bio-based projects. Institute for European Environmental Policy, AISBL

Benoît-Norris C, Traverso M, Valdivia S, Vickery-Niederman G, Franze J, Azuero L, Ciroth A, Mazijn B, Aulisio D (2013) The methodological sheets for subcategories in social life cycle assessment (S-LCA). United Nations Environment Programme and SETAC

Benoît-Norris C, Vickery-Niederman G, Valdivia S, Franze J, Traverso M, Ciroth A, Mazijn B (2011) Introducing the UNEP/SETAC methodological sheets for subcategories of social LCA. Int J Life Cycle Assess 16(7):682–690. https://doi.org/10.1007/s11367-011-0301-y

Calabrese A, Costa R, Gastaldi M, Levialdi Ghiron N, Villazon Montalvan RA (2021) Implications for sustainable development goals: a framework to assess company disclosure in sustainability reporting. J Clean Prod 319(128624). https://doi.org/10.1016/j.jclepro.2021.128624

Chhipi-Shrestha GK, Hewage K, Sadiq R (2015) "Socializing" sustainability: a critical review on current development status of social life cycle impact assessment method. Clean Techn Environ Policy 17(3):579–596. https://doi.org/10.1007/s10098-014-0841-5

D4EU (2022) Dendromass4Europe. About the project. https://www.dendromass4europe.eu/about-project

Dreyer LC, Hauschild MZ, Schierbeck J (2010) Characterisation of social impacts in LCA. Part 1: development of indicators for labour rights. Int J Life Cycle Assess 15(3):247–259. https://doi.org/10.1007/s11367-009-0148-7

Eberle U, Wenzig J, Mumm N (2022) Assessing the contribution of products to the United Nations" Sustainable Development Goals: a methodological proposal. Int J Life Cycle Assess 27(7):959–977. https://doi.org/10.1007/s11367-022-02063-8

European Commission (2012a) Innovating for sustainable growth: a bioeconomy for Europe

European Commission (2012b) Responsible research and innovation: Europe's ability to respond to societal challenges. Publications Office

Fürtner D, Perdomo Echenique EA, Hörtenhuber SJ, Schwarzbauer P, Hesser F (2022) Beyond monetary cost-benefit analyses: combining economic, environmental and social analyses of short rotation coppice poplar production in Slovakia. Forests 13(2):349. https://doi.org/10.3390/f13020349

Fürtner D, Ranacher L, Perdomo Echenique EA, Schwarzbauer P, Hesser F (2021) Locating hotspots for the social life cycle assessment of bio-based products from short rotation coppice. Bio Ener Res 14(2):510–533. https://doi.org/10.1007/s12155-021-10261-9

Global Bioeconomy Summit (2015) Making bioeconomy work for sustainable development

Global Risk Profile (2020) Global corruption index. https://risk-indexes.com/global-corruption-index/
von Grebmer K, Bernstein J, Wiemers M, Acheampong K, Hanano A, Higgins B (2020) Global hunger index. One decade to zero hunger. Linking health and sustainable food systems. Chatham House, Dublin/Bonn
Guinée JB, Heijungs R, Huppes G, Zamagni A, Masoni P, Buonamici R, Ekvall T, Rydberg T (2011) Life cycle assessment. Past, present, and future. Environ Sci Technol 45(1):90–96. https://doi.org/10.1021/es101316v
Heimann T (2019) Bioeconomy and SDGs: does the bioeconomy support the achievement of the SDGs? Earth's Future 7(1):43–57. https://doi.org/10.1029/2018EF001014
Herre B, Roser M (2016) Human rights. https://ourworldindata.org/human-rights
Herrera Almanza AM, Corona B (2020) Using social life cycle assessment to analyze the contribution of products to the sustainable development goals: a case study in the textile sector. Int J Life Cycle Assess 25(9):1833–1845. https://doi.org/10.1007/s11367-020-01789-7
Hesser F (2015) Environmental advantage by choice. Ex-ante LCA for a new Kraft pulp fibre reinforced polypropylene composite in comparison to reference materials. Compos Part B: Eng 79:197–203. https://doi.org/10.1016/j.compositesb.2015.04.038
Hetherington AC, Borrion AL, Griffiths OG, McManus MC (2014) Use of LCA as a development tool within early research: challenges and issues across different sectors. Int J Life Cycle Assess 19(1):130–143. https://doi.org/10.1007/s11367-013-0627-8
Ibáñez-Forés V, Bovea MD, Coutinho-Nóbrega C, de Medeiros HR (2019) Assessing the social performance of municipal solid waste management systems in developing countries: proposal of indicators and a case study. Ecol Indic 98:164–178. https://doi.org/10.1016/j.ecolind.2018.10.031
Iofrida N, De Luca AI, Strano A, Gulisano G (2018) Can social research paradigms justify the diversity of approaches to social life cycle assessment? Int J Life Cycle Assess 23(3):464–480. https://doi.org/10.1007/s11367-016-1206-6
ISO (2006) 14040:2006. Environmental management—life cycle assessment—principles and framework, 2nd ed, Switzerland
ITUC (2020) 2019 ITUC global rights index: the world's worst countries for workers
Jones A, Jakob M, McNamara J, Teutenberg A (2020) Review of the future of agriculture and occupational safety and health (OSH). Foresight on new and emerging risks in OSH. European Agency for Safety and Health at Work
Lettner M, Hesser F (2020) Asking instead of telling—recommendations for developing life cycle assessment within technical R&D projects. In: Albrecht S, Fischer M, Leistner P et al (eds) Progress in life cycle assessment 2019. Springer Cham, pp 173–188
Maier S, Beck T, Vallejo JF, Horn R, Söhlemann J-H, Nguyen T (2016) Methodological approach for the sustainability assessment of development cooperation projects for built innovations based on the SDGs and Life Cycle Thinking. Sustain 8(1006). https://doi.org/10.3390/su8101006
Mair-Bauernfeind C, Zimek M, Asada R, Bauernfeind D, Baumgartner RJ, Stern T (2020a) Prospective sustainability assessment: the case of wood in automotive applications. Int J Life Cycle Assess 25(10):2027–2049. https://doi.org/10.1007/s11367-020-01803-y
Mair-Bauernfeind C, Zimek M, Lettner M, Hesser F, Baumgartner RJ, Stern T (2020b) Comparing the incomparable? a review of methodical aspects in the sustainability assessment of wood in vehicles. Int J Life Cycle Assess. https://doi.org/10.1007/s11367-020-01800-1
Martin M, Røyne F, Ekvall T, Moberg Å (2018) Life cycle sustainability evaluations of bio-based value chains: reviewing the indicators from a Swedish perspective. Sustain 10(547). https://doi.org/10.3390/su10020547
Martínez-Blanco J, Lehmann A, Muñoz P, Antón A, Traverso M, Rieradevall J, Finkbeiner M (2014) Application challenges for the social Life Cycle Assessment of fertilizers within life cycle sustainability assessment. J Clean Prod 69:34–48. https://doi.org/10.1016/j.jclepro.2014.01.044
Mathe S (2014) Integrating participatory approaches into social life cycle assessment: the SLCA participatory approach. Int J Life Cycle Assess 19(8):1506–1514. https://doi.org/10.1007/s11367-014-0758-6

McManus MC, Taylor CM (2015) The changing nature of life cycle assessment. Biomass Bioener 82:13–26. https://doi.org/10.1016/j.biombioe.2015.04.024
Neugebauer S, Emara Y, Hellerström C, Finkbeiner M (2017) Calculation of Fair wage potentials along products' life cycle—introduction of a new midpoint impact category for social life cycle assessment. J Clean Prod 143:1221–1232. https://doi.org/10.1016/j.jclepro.2016.11.172
Niero M, Di Felice F, Ren J, Manzardo A, Scipioni A (2014) How can a life cycle inventory parametric model streamline life cycle assessment in the wooden pallet sector? Int J Life Cycle Assess 19(4):901–918. https://doi.org/10.1007/s11367-014-0705-6
Parent J, Cucuzzella C, Revéret J-P (2010) Impact assessment in SLCA: sorting the sLCIA methods according to their outcomes. Int J Life Cycle Assess 15(2):164–171. https://doi.org/10.1007/s11367-009-0146-9
Perdomo EEA, Ryberg M, Vea EB, Schwarzbauer P, Hesser F (2022) Analyzing the consequences of sharing principles on different economies: a case study of short rotation coppice poplar wood panel production value chain. Forests 13(3):461. https://doi.org/10.3390/f13030461
Perdomo EEA, Schwarzbauer P, Fürtner D, Hesser F (2021) Life cycle assessment of agricultural wood production—methodological options: a literature review. Bio Ener Res 14(2):492–509. https://doi.org/10.1007/s12155-021-10266-4
Pichler C, Fürtner D, Hesser F, Schwarzbauer P, Ranacher L (submitted) The role of the social Licence to operate in the emerging bioeconomy—a case study of short rotation coppice poplar in Slovakia
Pollok L, Spierling S, Endres H-J, Grote U (2021) Social life cycle assessments: a review on past development, advances and methodological challenges. Sustain 13(18):10286. https://doi.org/10.3390/su131810286
Ranacher L, Pollakova B, Schwarzbauer P, Liebal S, Weber N, Hesser F (2021) Farmers' Willingness to adopt short rotation plantations on marginal lands: qualitative study about incentives and barriers in Slovakia. Bio Ener Res 14(2):357–373. https://doi.org/10.1007/s12155-020-10240-6
Ritchie H (2022) What is undernourishment and how is it measured? Our World in Data. https://ourworldindata.org/undernourishment-definition. Accessed 8 April 2022
Rutz D, Janssen R (2014) Socio-economic impacts of biofuels on land use change. In: Rutz D, Janssen R (eds) Socio-economic impacts of bioenergy production. Springer International Publishing, Cham
Sachs JD, Lafortune G, Kroll C, Fuller G, Woelm F (2022) Sustainable Development Report 2022: from Crisis to Sustainable Development: the SDGs as Roadmap to 2030 and Beyond. Cambridge University Press, Bertelsmann Stiftung, Sustainable Development Solutions Network
Sala S, Vasta A, Mancini L, Dewulf J, Rosenbaum E (2015) Social life cycle assessment: state of the art and challenges for product policy support, 99101st edn. Publications Office of the European Union, Luxembourg (Luxembourg)
Siebert A, Bezama A, O'Keeffe S, Thrän D (2018a) Social life cycle assessment indices and indicators to monitor the social implications of wood-based products. J Clean Prod 172:4074–4084. https://doi.org/10.1016/j.jclepro.2017.02.146
Siebert A, Bezama A, O'Keeffe S, Thrän D (2018b) Social life cycle assessment: in pursuit of a framework for assessing wood-based products from bioeconomy regions in Germany. Int J Life Cycle Assess 23(3):651–662. https://doi.org/10.1007/s11367-016-1066-0
Spierling S, Knüpffer E, Behnsen H, Mudersbach M, Krieg H, Springer S, Albrecht S, Herrmann C, Endres H-J (2018) Bio-based plastics—a review of environmental, social and economic impact assessments. J Clean Prod 185:476–491. https://doi.org/10.1016/j.jclepro.2018.03.014
Sureau S, Neugebauer S, Achten WMJ (2020) Different paths in social life cycle impact assessment (S-LCIA)—a classification of type II impact pathway approaches. Int J Life Cycle Assess 25:382–393
Sutterlüty A, Šimunović N, Hesser F, Stern T, Schober A, Schuster KC (2018) Influence of the geographical scope on the research foci of sustainable forest management: Insights from a content analysis. Forest Policy Econ 90:142–150. https://doi.org/10.1016/j.forpol.2018.02.003
Svabova L, Metzker Z, Pisula T (2020) Development of unemployment in Slovakia in the context of the COVID-19 pandemic. Ekonomicko-manazerske spektrum 14(2):114–123. https://doi.org/10.26552/ems.2020.2.114-123.

Transparency International (2021) Corruption perception index 202. Berlin, Germany. https://www.transparency.org/en/cpi/2020/index/som. Accessed 8 March 2022
UN General Assembly (2015) Resolution adopted by the general assembly on 25 September 2015: transforming our world: the 2030 Agenda for Sustainable Development. United Nations
UNEP (2020) Guidelines for social life cycle assessment of products and organizations 2020. United Nations Environment Programme (UNEP)
UNEP (2021) Methodological sheets for subcategories in social life cycle assessment (S-LCA) 2021. United Nations Environment Programme (UNEP)
UNEP/SETAC (2009) Guidelines for social life cycle assessment of products. Guidelines for social life cycle assessment of product. United Nations Environment Programme, Paris
Weidema B, Goedkoop M, Mieras E (2018) Making the SDGs relevant to business. PRé Sustainability & 2.-0 LCA consultants
WHO (2021) Ambient (outdoor) air pollution. World Health Organization. https://www.who.int/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health
Wu R, Yang D, Chen J (2014) Social life cycle assessment revisited. Sustain 6(7):4200–4226. https://doi.org/10.3390/su6074200
Wulf C, Werker J, Zapp P, Schreiber A, Schlör H, Kuckshinrichs W (2018) Sustainable development goals as a guideline for indicator selection in Life Cycle Sustainability Assessment. In: Laurent A, Leclerc A, Niero M et al (eds) Procedia CIRP, 25th CIRP Life Cycle Engineering (LCE) Conference, Copenhagen, Denmark, 30 April–2 May 2018. 69th ed. Elsevir, pp 59–65
Yıldız-Geyhan E, Yılan G, Altun-Çiftçioğlu GA, Neşet Kadırgan MA (2019) Environmental and social life cycle sustainability assessment of different packaging waste collection systems. Resour Conserv Recycl 143:119–132. https://doi.org/10.1016/j.resconrec.2018.12.028
Zeug W, Bezama A, Thrän D (2021) A framework for implementing holistic and integrated life cycle sustainability assessment of regional bioeconomy. Int J Life Cycle Assess 26(10):1998–2023. https://doi.org/10.1007/s11367-021-01983-1
Zimmermann FM (2016). Was ist Nachhaltigkeit - eine Perspektivenfrage? In: Zimmermann F (Ed.). Nachhaltigkeit wofür? Von Chancen und Herausforderungen für eine nachhaltige Zukunft. Springer Spektrum, Berlin, Heidelberg, pp 1–24
Zira S, Röös E, Ivarsson E, Hoffmann R, Rydhmer L (2020) Social life cycle assessment of Swedish organic and conventional pork production. Int J Life Cycle Assess 25(10):1957–1975. https://doi.org/10.1007/s11367-020-01811-y

Chapter 8
Life Cycle Sustainability Assessment for Sustainable Bioeconomy, Societal-Ecological Transformation and Beyond

Walther Zeug, Alberto Bezama, and Daniela Thrän

Abstract Decoupling the fulfillment of societal needs from an ever-increasing production of goods together with decoupling this sufficient production from negative environmental, social and economic impacts, is and will be the major challenge of our economic systems to avoid an even deeper socio-ecological crisis. The ascending bioeconomy practices have to be assessed with regard to their potential to provide a good life for all within planetary boundaries Addressing this, life cycle sustainability assessment (LCSA) is necessary to integrate social, environmental and economic sustainability assessments. However, LCSAs are still in their infancy and a series of practical problems can be traced back to a lack of sound sustainability concepts and applied political economy/ecology. We reflect on social, ecological and economic sustainability, our societal relations to nature and a necessary societal-ecological transformation in order to structure a systemic framework for holistic and integrated LCSA (HILCSA). This framework allows an implementation in openLCA, conducting the inventory and impact assessment with harmonized databases and more coherent results compared to previous approaches. For further development we identify questions of political economy/ecology as significant. The idea of a bioeconomy as well as systemic assessments is a question of the perception of ends and means of a societal transformation.

Keywords Life cycle sustainability assessment · Bioeconomy · Political economy · Decoupling

W. Zeug (✉) · A. Bezama · D. Thrän
Department of Bioenergy, Helmholtz-Centre for Environmental Research (UFZ), 04318 Leipzig, Germany
e-mail: walther.zeug@ufz.de

D. Thrän
Department Bioenergiesysteme, Deutsches Biomasseforschungszentrum Gemeinnützige GmbH, Torgauer Str. 116, 04347 Leipzig, Germany

F. Hesser et al. (eds.), *Progress in Life Cycle Assessment 2021*,
Sustainable Production, Life Cycle Engineering and Management,
https://doi.org/10.1007/978-3-031-29294-1_8

8.1 Preliminary Considerations on Implicitly Underlying Concepts

8.1.1 Sustainability Concepts and (Bio)Economy Under Different Paradigms of Capital

The ecological challenges our global societies face are not only related to climate change, as it is likely that humanity is about to cross several planetary boundaries (PB)—representing the ecological limits of our planet—with feedbacks difficult to handle and partly irreversible (O'Neill et al. 2018; Rockström et al. 2009; Steffen et al. 2018). Practically no country performs well on both the biophysical and social dimensions, being the general rule that when the more social needs are achieved, the more biophysical boundaries are transgressed, and vice versa (O'Neill et al. 2018). For example, Germany's environmental footprint is 3.3 times higher than its biocapacity (Bringezu et al. 2020; GFN 2019; Network 2019; Schaefer et al. 2006). Fulfillment of societal needs is seemingly directly coupled with transgressing PB (Haberl et al. 2012; O'Neill et al. 2018).

As one way to address these challenges more than 50 countries worldwide have now developed bioeconomy (BE) related policy strategies (Bell et al. 2018; German Bioeconomy Council 2018b; Kleinschmit et al. 2017; Meyer 2017) to achieve sustainable development, depending on how this is understood in the respective strategies. BE is broadly understood as "the production of renewable biological resources and the conversion of these resources, residues, by-products and side streams into value added products, such as food, feed, bio-based products, services and bioenergy" "within the framework of a sustainable economy" (German Bioeconomy Council 2018a). However, there is and most probably will be no unified definition of BE (Birner 2018), since different and partly contradicting interest groups (Bioökonomierat 2022; Meyer 2017; OECD 2018) and diverse social mentalities result in conflicts (Eversberg and Fritz 2022; Zeug et al. 2019), e.g. bioeconomy as a technological solution to enable further growth in 'green capitalism' vs. bioeconomy as a socio-ecological transformation. Nevertheless, a common approach can be to see BE as part of a social-ecological transformation to address global challenges of the twenty-first century (Bioökonomierat 2022).

Sustainability as a state, or more precisely sustainable development (SD) as a process, is often attributed to meeting the needs of the present without compromising the ability of meeting needs in the future (Brundtland et al. 1987). Economic growth to reduce poverty was the specific sense of a solution conferred to, and, in doing so, to create the wealth, technology and commitment necessary to reduce ecological damage. The terms SD and sustainability are often used synonymously, although SD is based on a dualist anthropocentric view that humankind has a special and almost detached relationship with nature and is only interested in the instrumental or utilitarian value attached to an ecosystem (shallow ecology). Resources should be managed to be available for future generations, natural and human capital are interchangeable and nature should be cared about only to the extent considered as human interests (Hector et al. 2014). This results in a dualism of humankind and nature with a clear hierarchical order that humankind rules over nature (Görg 2004). On the other hand, (strong) sustainability strives for some form of dynamic equilibrium in which the needs of humankind and the needs of nature are both satisfied. In a broader notion of environmental-preservationist this means that the natural world ought to be preserved and must not be allowed to deteriorate, disappear or be dominated by humans (deep ecology). Here humanity is an integral part of nature, not separated from it, and nature has an intrinsic value (Hector et al. 2014; Mebratu 1998). This polarized constellation of anthropocentric (weak sustainability, shallow ecology, SD) and ecocentric (strong sustainability, deep ecology) views is an epistemological trap: the two positions are permanently irreconcilable and based on different self-evident axioms (Hector et al. 2014; Zeug et al. 2020) (Table 8.1).

These discourses, mostly implicitly, shape understandings of (bio-)economy and sustainability assessment methods today: On the one hand, neoclassical environmental economics are associated with weak sustainability because they clearly possess an anthropocentric concept of SD, characterized by 'benefit and welfare', which in capitalism is synonymous with profit maximization. It is assumed that natural capital can be substituted with artificial capital, the environment is frequently undervalued, tends to be overused and if the environment only were given its 'proper value' in economic decision-making terms, it would also be protected much more highly (Hector et al. 2014; Mebratu 1998; Redclift and Benton 1994). But even within neoclassical models, this constant substitutability of capital stocks, the timely availability of innovations and backstop technologies (enable the use of resources for an indefinitely long time) like BE allow the assumption of non-existent growth limits, without depleting non-renewable and overuse renewable resources (Bennich and Belyazid 2017; Smulders 1995). Thus, unlimited economic growth is only possible if enough human capital is allocated to R&D to sufficiently increase the necessary efficiency of resource use without necessitating fundamental changes (Barbier 1999; Michel and Rotillon 1995; Perdomo Echenique et al. 2022; Verdier 1995; Victor et al. 1994). This points to why there is such a mainly technological focus on BE

Table 8.1 Contents of popular sustainability concepts (Hector et al. 2014; Hopwood et al. 2005; Mebratu 1998; Ramcilovic-Suominen and Pülzl 2018)

Keywords	Shallow Ecology Weak sustainability Prudentially-conservationist Anthropocentric Sustainable development	Deep ecology Strong sustainability Environmental-preservationist Ecocentric Sustainability
Content	Humanity with specific relation towards nature, instrumental value of ecosystems, positivist view, mechanistic systematization, substitutability of capitals, objective: economic sustainable development	Humanity as integral part of nature, intrinsic value of ecosystems, monist and morally egalitarian view, preservation of nature and non-substitutability, objective: sustainable equilibrium

and in most sustainability assessments. With that come conceptual and methodological shortcomings: tending to overlook or deliberately reject the relevance of non-human species, tending to be mechanistic and reductionist about society, ecology and economics (Hector et al. 2014). Consequentially, sustainability assessments not only tend to treat environmental problems without tackling the underlying causes and assumptions that underlie our current political and economic thinking (Mebratu 1998), but also to see social, environmental as economic aspects and sustainability as rather detached from each other. As a result, approaches develop which are non-integrative and additive that entail explicit or implicit positivism. From a positivistic perspective, reality is seen as independent, objective, empirical and measurable; there are general laws between variables representable by mathematics; methods are model simulations, manipulation of variables and quantitative data; and governance or policymakers 'outside' the system have to pull 'levers' to steer developments.

On the other hand, there is an interdisciplinary and more qualitative concept of ecological economics tending towards strong sustainability (Georgescu-Roegen 1971). In this time and context of ecological economics the term 'bioeconomics' occurred for the first time, but had a completely other meaning than the current term of BE (Birner 2018): the earth is seen as a closed system in which the economy is a subsystem and, therefore, there are limits to resource extraction; a sustainable society-wide system with a high quality of life of all inhabitants within the natural limits is sought; complex systems are of great uncertainties and require a preventative approach; a fair distribution and an efficient allocation are necessary (Costanza et al. 1997; Hauff and Jörg 2013). In terms of sustainability assessment, a consequence is to consider PB as absolute limits of resource extraction. In contrast to pursuing individual gain, benefit and profit maximization, the ecological economy is strengthening the importance of ecological systems for the safeguarding or improvement of societal conditions. In other words, it is about the welfare of the whole society (Hauff and Jörg 2013). In particular, the assumption of substitutability of natural and artificial capital is called into question, since human capital is needed to make efficient use of natural capital, and natural capital is needed to generate anthropogenic capital (Hauff and Jörg 2013; Hector et al. 2014). Capitals are indeed substitutable, but any number of workers and machines or an increase in productivity cannot completely replace the starting materials necessary for production. A necessary increase in productivity can be achieved through three approaches relevant for the BE and their restrictions: increasing the flow of natural resources per unit of natural capital, limited by biological growth rates; increasing product output per unit of resource input, limited by mass conservation; increasing efficiency of use of conversion of raw materials into products, limited by technology (Costanza et al. 1997).

In the currently dominant neoclassical ideology, BE is interpreted as both: a variable production factor technology as well as additional natural resources to be used for additional growth. The notions and political BE discourses in the EU were dominated by biotechnology visions from industrial stakeholders (Hausknost et al. 2017; Staffas et al. 2013). Therefore, BE was mainly seen as the appropriate endogenous technology factor and immediate precursor in the neoclassical concept of SD by providing sufficient resources and using them to increase benefit and profit maximization, which set the stage for the win–win–win narrative of the BE (Kleinschmit et al. 2017). Biotechnology in this sense would likely raise further huge sustainability risks when it is upscaled to an industrial level, as it is already, and will absorb large-scale biomass flows demanding significant exports and imports (Bringezu et al. 2020; Budzinski et al. 2017; Gawel et al. 2019). A growing BE in Europe has already led to an increase in harvested forest area and imported biomass and may hamper forest-based climate mitigation (Erb et al. 2022; Palahí 2021). These aspects may be a reason for the still low public 'acceptance' or explicit criticism of the BE (Mustalahti 2018; Stern et al. 2018) and that the majority of NGOs have a rejecting perspective on BE as a PR campaign from industrial business to green-wash their business as usual (Gerhardt 2018; Šimunović et al. 2018). Nevertheless, a climate-neutral economy will depend on these enormous material flows of sustainable and renewable biomass. The techno-political option space of the BE (Hausknost et al. 2017) shows strong connections to the presented sustainability and economy concepts: "Sustainable Capital" corresponds to the neoclassical perspective and weak sustainability, as well as, "Eco-Growth" corresponds to the ecological economics perspective and weak sustainability as to forms of ecological modernization; "Eco-Retreat" is more an ethical vision of deep ecology, strong sustainability and ecological economics; "Planned Transition" is based on ecological economics but neither corresponds clearly to weak nor strong sustainability and will be important in the following (Zeug et al. 2020).

8.1.2 Sustainability and LCSA

Measurement and evaluation of so called ecological, economic or social sustainability at different scales is the central motivation of different methodological frameworks of life cycle assessments (LCA) and their combination or integration in life cycle sustainability assessments (LCSA). Especially the latter methods of LCSA are still at an early stage and face significant methodological problems (Guinée 2016; Ingrao et al. 2018; Zimek et al. 2019). Comprehensive reviews of LCSA approaches identify the lack of transparent description and discussion about implicitly underlying concepts of sustainability, and resulting difficulties in the classification of indicators and criteria as major obstacles (Wulf et al. 2019). At least there are currently two definitions of LCSA (Sala et al. 2012a, b). On the one hand, the widely used and highly operationalizing and *additive* scheme (LCSA = ELCA + LCC + SLCA), first proposed by Klöpffer in 2008 (Kloepffer 2008). It argues that on the basis of the three-pillar approach, the three methods of environmental-LCA (ELCA), social-LCA (SLCA) and life cycle costing (LCC) have to be standardized, harmonized, synchronized (mostly this means an analog brief structure as in DIN EN ISO 14040 and 14,044) (Valdivia et al. 2021) and then combined, whereas extensive qualitative analyses are excluded. On the other hand, there is at least the idea of an *integrative* approach first proposed by Guinée in 2011 (Guinée et al. 2011), where within a common sustainability concept and methodical framework impact categories from E-LCA, S-LCA and LCC should be integrated into a holistic assessment. However, as recent comprehensive reviews (Costa et al. 2019; D'Amato et al. 2020; Fauzi et al. 2019; Troullaki et al. 2021; Wulf et al. 2019; Zimek et al. 2019) show: nearly all LCSA approaches more or less follow the additive scheme and are explicitly or implicitly based on the three-pillar-approach (Zimek et al. 2019) with respective consequences.

The so-called three pillar approach (people, planet and prosperity) of the World Summit on SD in 2002 has prevailed and is essential to the present understandings of sustainability (Elkington 1998; Hector et al. 2014; UNEP 2011). In the updated guidelines for S-LCA, prosperity is even directly identified with profit (UNEP 2020). Thereby suggested are kinds of several more or less differentiated entities constituting sustainability in a complementary and constructive way (Meadowcroft 2007). The most established and used resulting model (see Fig. 8.1, left) from the three pillar approach is the reductionist model of interlinked systems (Holmberg et al. 1992) as the dominant model (cf. Rockström and Sukhdev 2016). However, it leads to inflexible and polarized oppositions due to its reductionist epistemological foundations of ecological vs. social vs. economic, and oftentimes some kind of equilibrium or viable and equitable state is considered as sustainability in the center or when dimensions are overlapping (Elkington 1998; Redclift and Benton 1994; Trzyna et al. 1995).

Additive LCSA takes the three parts respectively dimensions of sustainability as the point of departure (Fig. 8.2) and considers LCSA likewise as a linear summation and combination of the parts: E-LCA, S-LCA and LCC are carried out more or less independently from each other as separate systems (Fig. 8.2c). Broadly said, scopes, corresponding methods and indicators of the life cycle inventory (LCI), life cycle

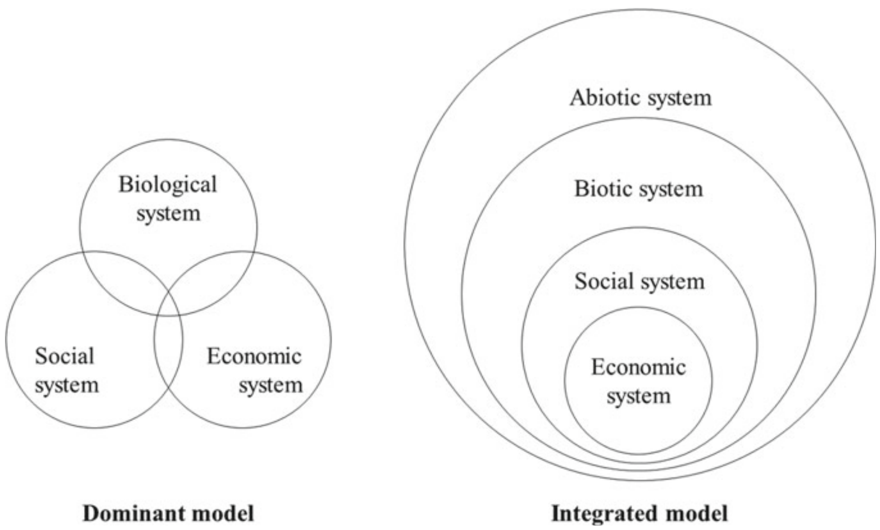


Fig. 8.1 Schemes of sustainability concepts, adopted from (Mebratu 1998, Fig. 1)

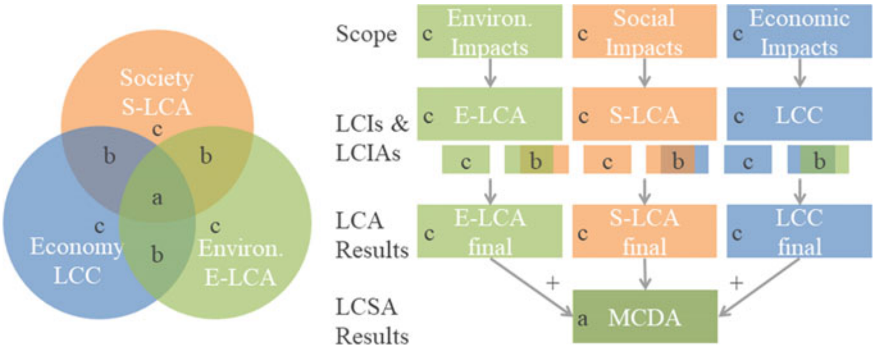


Fig. 8.2 Three-pillar-approach of sustainability and additive scheme of LCSA = ELCA + LCC + SLCA, (**c** separate systems, methods and indicators, **b** intersection between two systems, indicators which cannot be clearly assigned to one system, **a** all dimensions somehow combined, additive combination of methods results; LCI—Life Cycle Inventory, LCIA—Life Cycle Impact Assessment)

impact assessment (LCIA) as well as their individual results only have in common that they relate to the same product or functional unit which is to be assessed (cf. Ekener et al. 2018; Suwelack 2016; Urban et al. 2018). When assigning the indicators to impact categories, and/or when indicators are allocated to sustainability dimensions, it becomes apparent that for some indicators no clear intuitive allocation is possible or useful (e.g. aspects like sustainable final consumption/production, infrastructures, development of rural areas, employment (Egenolf and Bringezu 2019)). Such aspects mostly describe complex relations between two or more sustainability dimensions and are not even roughly categorizable as solely social, economic or ecological (b).

Dealing with such issues is difficult within the three-pillar-approach and separate assessment methods, since a simple combination of the particulate methods is only possible to a very limited extent (Costa et al. 2019; Keller et al. 2015; Wulf et al. 2019) and combining the final results with MCDA (Ekener et al. 2018; Sala et al. 2012a) does not represent an integration of social, ecological and economic aspects. The analysis of complex systems by their subsystems would mean more than just combining their parts (Halog and Manik 2011). Such process-based approaches with a high technical detail but few general preliminary considerations result in a series of specific problems occurring in operationalization at the latest: trade-offs and conflicts of objectives (Guinée 2016), double-counting and problems of monetization (Guinée 2016), allocation to fuzzy impact categories (e.g. if an indicator is of primarily social, environmental or economic character or which stakeholders are effected), functional units (Costa et al. 2019), exogenous and endogenous weightings in accounting (Traverso et al. 2012), rating, normative goal systems and many more. For instance, the decoupling debate has shown that improving the ecological performance of products only has a limited effect on global environmental challenges, and pareto effects come to bear which makes a relatively small number of causes responsible for a major portion of the effects, resulting in a need for hot spot analyzes (Halog and Manik 2011). Generally speaking, a theoretically well-founded and holistic social, ecological or economic sustainability theory from political economy and political ecology is missing in LCSA. Integration would mean, considering social, ecological and economic aspects as one system, and holistically thinking about the transdisciplinary contextualization of LCSA in social and political science (see Sect. 8.2). In the ongoing discussion of the last years, a broad spectrum of blended approaches emerged (de Schutter et al. 2019; Liu et al. 2015; Purvis et al. 2019; Sala et al. 2012b). However, there is another rather less-established model of integrated systems in accordance with ecological economics (see Fig. 8.1, right) (Mebratu 1996). Presumably rather less-established, since its theoretical conception is less intuitive and requires a well-founded theory, as well as its practical implications are far stronger. In the following, we will introduce a founded theory to employ this concept in models of sustainability assessment, in particular LCSA.

8.2 Introduction of Critical Concepts for Progress in LCSA

8.2.1 Transdisciplinarity

Our previous considerations already show the importance of implicitly underlying social science and economics and how they influence LCA and LCSA approaches. Consequently, the need for a transdisciplinary sustainability science aiming at understanding interactions between nature and society has often been stated in the literature for LCSA (Sala et al. 2012a, 2012b), but rarely substantiated or implemented (Future Earth 2016; Pfau et al. 2014). A lot of knowledge and evidence of relationships (e.g. between SD and climate action) are scattered across different institutions, locations and disciplines; this fragmentation is a critical barrier to a holistic and integrated understanding of social, economic and environmental systems (Knierim et al. 2018; Nerini et al. 2019). The methods and findings of different scientific disciplines are oftentimes very rational, competent and innovative within their respective fields of expertise, but neglect or contradict insights from other disciplines and are embedded in possibly irrational frameworks or ideologies (Demirovic 2003). We understand interdisciplinarity as an exchange and dialogue between disciplines, whereas transdisciplinarity as a research paradigm of sustainability sciences aims for holistic thinking: an inherent contextualization and embedding of findings within a greater context creating transcending insights (Klein 2008; Knierim et al. 2018; Lubchenco et al. 2015). Real-world problems are the starting point of transdisciplinary research, to gain a better understanding of social-ecological problems and contributing to their solution is the research objective (Jahn et al. 2012; Kramm et al. 2017). Of course, modern science is much too complex to be covered by one person and so transdisciplinary practice means at least working together, recognizing each other and involving stakeholders to develop novel conceptual and methodological frameworks with the potential to produce transcendent theoretical and practical approaches (Hummel et al. 2017; Klein 2008; Rosenfield 1992). The resulting methodological pluralism can lead to more consistency and less bias (Lamont et al. 2006). Attributes like 'social' and 'economic' do not describe separate objects of scientific observation, but rather different perspectives on the same objects and the underlying relations. Transdisciplinary means to understand and reflect a seemingly ecological research question as a simultaneously political-economic research question and vice versa. Consequentially, ecological arguments can never be neutral any more than sociopolitical or economic arguments are ecologically neutral (Harvey and Braun 1996). This means that for achieving a sustainable transition to a BE, there is not only a need to transform so called societal and industrial mindsets, and not only a question of a few 'tweaks' to the system. Instead, it is actually a question of transformations of our very fundamental societal relations to nature (SRN) (de Besi and McCormick 2015; Kramm et al. 2017; Pichler et al. 2020). Different means, ends, and values seem to be the guiding factors in what we have understood as conflicting interests and perceptions in BE assessments (Zeug et al. 2019). Simply setting ambitious goals, but ignoring ideologies, social norms and values, religious beliefs and institutions, including formal and informal rules and customs will not be sufficient (Norström 2013; Stegemann and Ossewaarde 2018). Only technological changes and innovations, a sole focus on industrial efficiency or simply replacing fossil resources with biomass are in danger of maintaining the same production and consumption system as the fossil-based economy (de Besi and McCormick 2015). Such insights go back to early interdisciplinary materialism, later critical theory, and social ecology are applied to the concept of SRN. They reveal that there is no non-normative science; if there is no explicit scientific value judgment there is an implicit one confirming the status quo (Amidon 2008; Hummel et al. 2017; Kramm et al. 2017). Regarding progress in LCSA, the following framework aims for embedding positivist data-driven methods of science into a relativist and postmodernist philosophy of science, combining the strengths of quantitative systems modeling as well as political economy and ecology. Even though this is and will remain a field of tension (Bauriedl 2016), due to the complexity and different perspectives of methods. Transdisciplinarity is, therefore, necessary to achieve a proper integration of methods in an LCSA. As well on a regional scale, transdisciplinary approaches offer new possibilities of deliberative methods to find normative constellations of societal needs through stakeholder participation (e.g. interviews and discussions).

8.2.2 Societal Relations to Nature

As shown, none of the dualistic approaches alone is sufficient, neither anthropocentric nor ecocentric, neither weak nor strong sustainability, and especially not the dominant and reductionist model of sustainability. But rather the integrated model and a corresponding holistic thinking based on the interactions and relations between the parts and the whole. Therefore, we take up the concept of SRN towards a holistic and integrated LCSA (HILCSA). In SRN nature, economy and society do not stand in an external relation to each other nor do they exist by themselves as the three-pillar approach suggests, rather, they constitute each other through their relations (Görg 2003, 2011; Görg et al. 2017; Hummel et al. 2017; Kramm et al. 2017; Pichler et al. 2020, 2017):

The SRN concept at its core evolves around the idea of societal needs and SRNs should be regulated to fulfill them. Thus, SRN is not only complementary and a well-founded theory for the SDGs, but also incorporates the concept of provisioning systems, justice (Menton et al. 2020), equity, and critically reflected SD. Social ecology and SRN conceptualize societies as simultaneously subject to biophysical and socio-cultural spheres of causation in a social metabolism. Nature and society are different things, and although distinct, not independent from one another. What nature is results from what society, culture, technology, etc. is not, and vice versa. Social metabolisms transform a society's energetic and material inputs, integrate them into societal stocks or other socio-economic systems, and discharge them to the environment as wastes and emissions. Industrial and BE metabolisms are special cases of social metabolisms (Bezama et al. 2021). However, this societal metabolism has no essential or eternal nature (Pichler et al. 2017). Instead historically, geographically and culturally specific socio-cultural mechanisms like politics and economic patterns are in place through which a society organizes its metabolism. In general, our SRN are shaped by economies, which are temporally and geographically different (e.g. transformable) social systems supposed to satisfy societal needs (ends) utilizing natural resources, labor and technologies (means). Especially important for LCSA are working hours as the crucial (activity) variable in production processes, since labor is not only the origin of economic value but as well relates social effects to production processes (Fröhlich 2009; Postone 1993).

These economic, and therefore also societal, mechanisms are understood as specific patterns of regulation, and fail when interactions with nature become dysfunctional, e.g. overexploitation of natural resources (overfishing, deforestation, soil degradation) or failure of a mechanism for effective and efficient allocation (hunger, poverty). Although there is the idea of being able to dominate nature, and nature is increasingly shaped by societal activities, it is becoming increasingly clear that global societies are significantly affected by environmental impacts and crisis trends. In this regard, we speak of the *Capitalocene* instead of the Anthropocene (Brand and Wissen 2018), since capitalism as the currently dominant societal and economic system has led to a social-ecological crisis, and not humankind itself as the term Anthropocene suggests. In specific our SRN are shaped by capitalism as a historically specific form of economy: a societal system that perpetuates the growth and accumulation of value (end) through societal needs using natural resources, labor and technologies (means). The fulfillment of societal needs is not the purpose of capitalist economic activities, but as well a necessary mean as all other production factors are to gain profit (Postone 1993). But why is the production of raw materials, resource consumption and negative impacts growing and need to grow too? In 'capital-ism' the imperative of capital accumulation, growth and the predominance of the production of surplus values over the production of use values is dominant (Postone 1993). Societal needs (use value) are only satisfied if they are coupled with sufficient purchasing power (exchange value). Both values use and can overuse resources, but monetary or exchange values tend to ignore the biophysical requirements of ecosystems categorically, e.g. externalities like environmental degradation are not intended to be internalized (Schleyer et al. 2017). Since the exchange value of commodities and money is the starting and the end point of every capitalist economic process, profit becomes the main driver and end in itself. If everything depends on an abstract quantitative value, the only driver is the endless growth of this value, and consequentially there is no "enough". Exchange value in the long term depends on the use value and production of material commodities, leading to valorization and overexploitation of natural and human capital and likewise growing negative social impacts and transgression of PB. Solely new technologies like BE in green capitalism as the potential of additional growth usually expand and/or shift the exhaustion of one resource to another. Growth in GDP (exchange value) ultimately cannot plausibly be decoupled from growth in material and energy use (use value), therefore, GDP and material growth cannot be sustained infinitely in this very economic system (Zeug et al. 2019, 2021b). Beyond that, a significant increase in labor productivity through automatization and digitalization leads to exponentially growing economic material output but stagnation and even a decrease in GDP per capita, profit rates, real loans and equality, especially in affluent and industrialized countries (Brynjolfsson and Andrew 2015; Piketty 2014). But also globally the labor's share of GDP had declined since there is a tendency toward higher capital productivity in capital than in labor and so shifting the investments from labor to capital (Karabarbounis and Neiman 2013). When growing economic production is not decoupled from its ecological impacts, but income and affluence are decoupled from this very production, then a good life for all within PB will be hard to achieve when income is a prerequisite for achieving nearly all societal needs.

A good example of capitalist SRN and patterns of regulation is the apparent connection between ending poverty (SDG 1) and ending hunger (SDG 2), both considered by stakeholders as very relevant for the BE (Zeug et al. 2019). In this case, even if enough food is produced worldwide to end hunger, the pattern of regulation of our economies requires ending poverty first. Since societal needs alone (use value), sufficient resources and means do not lead to their fulfillment, as long as those needs and preconditions are not coupled with enough purchasing power and income (exchange and surplus value). The same is true for the fuel vs food debate in BE: land or crops will be used for the purpose with the highest expected surplus value (e.g. fuels), instead of the fulfillment of more basic societal needs with a higher use but lower exchange value (e.g. nutrition) (cf. Ashukem 2020).

8.2.3 Societal-Ecological Transformation

Transformations take place as changes in initial patterns of regulation to new ones when the old ones become dysfunctional (Wittmer et al. 2022). The role of power relations in enabling and maintaining unsustainable resource use patterns, the role of social-ecological innovations within transformation processes and transregional interdependencies have been identified as emerging clusters of challenges in societal metabolism (Pichler et al. 2020). Terms and concepts of transformations toward sustainability remain fuzzy and there is much ambiguity and disagreement about the meaning and function of these concepts (Görg et al. 2017). Such transformation will have to innovatively address normative and socio-economic barriers, like global political patterns of regulation and resulting production and consumption patterns, as well as technological and ecological challenges. For example, technological inventions must go hand in hand with social, economic and organizational innovations, and questions of scale arise in the field of tensions between a global socio-ecological crisis and the responsibility and scope for action at the local and regional levels.

A potential future societal-ecological transformation should incorporate the PB as the main ecological limits, e.g. a certain GHG concentration should not be exceeded as well as there is a limit for the use of land, resources, water and so on (O'Neill et al. 2018). PB are not necessarily constant over time and nor a deterministic constant, but at least most likely are scenarios in which the transgression of one PB leads to even more transgressions of other PB Rockström et al. 2009; Steffen et al. 2018), e.g. climate change induces water scarcity and land degradation. In difference to common concepts of PB, from a perspective of political ecology, PB should be understood as socially constructed and politically contested (Bauriedl 2016; Görg 2015). As a qualitative simplification, we assume the PB as constant (Fig. 8.3) and their transgression as to be avoided.

Displayed as qualitative trends derived from quantitative charts (Roser 2022), ecological impacts and resource use grew and grow exponentially, especially since the 1950s and temporarily are exceeding PB globally by far. Whereas the production of material and immaterial commodities (e.g. GDP) as the cause for transgressing PB increases even more exponentially (ibid.) (cf. Sect. 8.2.2). However, the development of social indicators like the human development index rather has a far less exponential and more linear trend. This not only illustrates the production of exchange values by commodities as a main driver of production, resource use and environmental impacts in capitalism, but as well the quality in which societal needs are disproportionally coupled to commodity production. However, these qualitative trends correspond more to industrialized countries of the global north and negative impacts are shifted especially to the global south (Bauriedl 2016; Görg 2015).

A societal-ecological transformation would have to change patterns of regulation and societal relations in a way which, in technical terms, can be described as double decoupling: a societal as well as a techno-economic decoupling, which are mutually dependent and related to each other. On the one hand, the societal decoupling would

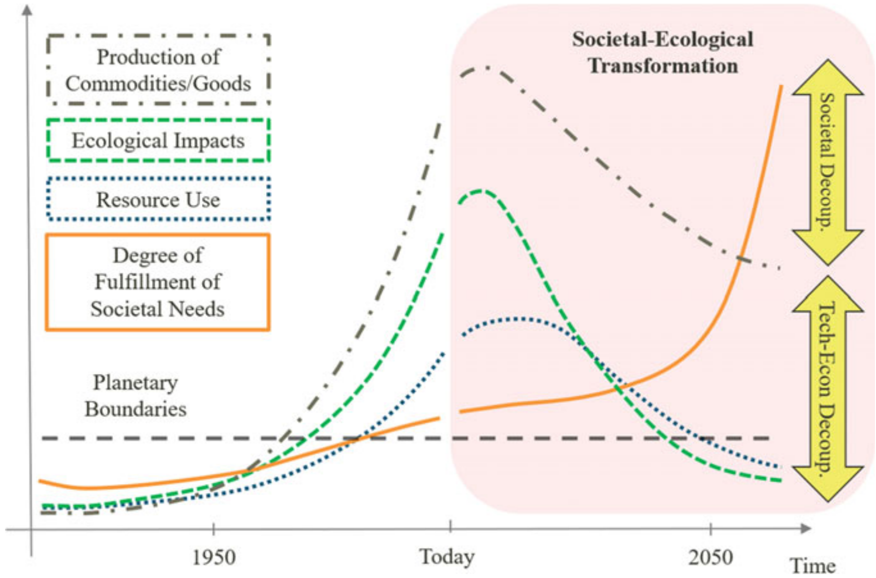


Fig. 8.3 Societal-ecological transformation and double decoupling as qualitative trends

have to decouple the degree of fulfillment of societal needs from an increasing production of material goods and overcome their commodity character, e.g. sufficiency. Such a societal-ecological transformation on a societal level means mainly a reconsideration of the economy as a satisfaction of societal needs (ends) by means of natural resources, labor and technologies (means). Innovation and sustainable technologies alone will not solve this predominantly political challenge. This does not mean that there is a contradiction between substitution and innovation. On the contrary, innovation is one of the prerequisites for substitution. Beyond economic substitution, for most of the biophysical–social indicator linkages diminishing marginal utilities were identified: from a certain degree of affluence and fulfillment of societal needs every additional unit of resource use contributes less to social performance, making sufficiency an essential factor for economic sustainability (O'Neill et al. 2018). Without a societal decoupling there is relative decoupling (fewer impacts per product, techno-economic) but no absolute decoupling (fewer impacts in total, societal), absolute decoupling is not plausible in a growing economy. LCSA in this regard can provide some information by the following dimension.

On the other hand, the techno-economic decoupling means decoupling the remaining sufficient and necessary material production from increasing resource use and negative ecological, social and economic impacts. A BE and circular economy (D'Amato 2021) will be decisive but are not sustainable per se and therefore LCSA can make significant contributions for sustainability assessments. Sustainable BE has to be a highly effective (fulfills societal needs), efficient (achieving most with less) and just (nobody falls behind) use of renewable resources within PB. Unique about the BE provisioning system is its inherent capacity for regeneration, allowing natural or biological resource stocks to replenish after extraction, and they are typically in constant interaction with their surrounding systems (Erb et al. 2022; Lindqvist et al. 2019; Zörb et al. 2018). Whereas every unit of non-renewable resources used now is a resource which will not be available in the future and thereby comprises intra- and intergenerational equity (Fedrigo-Fazio et al. 2016; Parrique 2019). But BE as industrial metabolism is only sustainable if: the rate of extraction does not exceed the rate of regeneration; the regenerative capacity is not diminished by extraction, processing, and utilization of resources; material and energy cycles are increasingly linked; and societal needs are fulfilled as well as they are the central objective of the economy itself. In contrast to non-renewable fossil systems, these complex interactions make the management of the BE complex and require fundamentally different strategies of planning (Erb et al. 2022; Lindqvist et al. 2019). The main limiting long-term factors of BE is the conversion efficiency of 1–2% of plants turning sunlight into carbon; and the limited areas where the sun shines, sufficient water is available and plants can grow without causing negative feedbacks like accelerating forestry erosion, soil erosion or biodiversity loss. Besides, the concept of reduce, reuse and recycle can actually be put into practice in the right order, since today a reduction or sufficiency of production and consumption is incompatible with the imperative of growth.

Hence, a societal-ecological transformation and sustainable BE corresponds strongly to the "Planned Transition" techno-political vision of BE (Hausknost et al. 2017). This means that on the one hand advanced technologies on a large-scale industrial level (integrated biorefineries, cascade use, eco-functional intensification of certain agricultural sectors, global trade in certain biogenic commodities, use of high-tech biotechnologies) will be needed to achieve the very ambitious demands on resource efficiency (Aguilar et al. 2018; Nitzsche et al. 2016; Olsson et al. 2016; Panoutsou et al. 2013). On the other hand, further growth, capital accumulation and an invisible hand are not a necessary part of BE. Rather, not transgressing the PBs, fulfilling essential societal needs and socially conscious planning of this transformation are.

8.3 Holistic and Integrated Life Cycle Sustainability Assessment

The framework of HILCSA aims to take the previously discussed complex problems into consideration, as far as it is possible in a broad understanding of LCSA methods. Holistic in this regard means to have the bigger picture in mind: not only to have a transdisciplinary and critical background theory of political economy, but as well to not fall short on the implications which some of the results may have and impose fundamental societal transformations, instead of only technological innovation or doing some 'tweaks in the system'. Whereas integrated stands for an integrated model of sustainability (cf. Fig. 8.1) which enables redeeming the integrated approach suggested by Guinee et al. (2011): to integrate social, ecological and economic sustainability assessment into one unified method instead of additionally combining different methods (see Sect. 8.1.2).

First, the spatial and temporal level of LCSA in general and HILCSA in particular, which deals with social-ecological transformations and SRN, is the mesolevel of economic organizations and institutions as actors of industrial metabolism. Besides, there are micro levels of individual actions and macro levels of societal powerful patterns of regulation. On this meso scale, HILCSA is in particular useful to assess techno-economic and relative decoupling, and needs to at least be aware of implications and relations of the micro and macro scale, or embedded in a transdisciplinary framework. We deem the three-pillar approach as not suitable for an integrated and holistic LCSA as well as a cause of major methodological problems (Sect. 8.1.2). Instead, we propose an integrated sustainability framework filling the identified research gap of a missing framework for HILCSA (Fig. 8.4i). Second, in contrast to the additive LCSA (LCSA = S-LCA + E-LCA + LCC), the HILCSA (HLCSA = f (S-LCA, E-LCA, LCC)) framework builds on this integrated sustainability framework for operationalization and integrates social, economic and ecological aspects in a common goal and scope, LCI, LCIA, results and interpretation (Fig. 8.4ii).

Economic systems on a meso scale are handled as product- and process-systems in LCA, comprising both physical and social systems, mediating the relationship between natural resources and societal needs through economic infrastructures and practices. When normatively aiming at a good life for all within planetary and regional boundaries, an integrated sustainability model puts social, ecological and economic sustainability in a specific relation: SRN which fulfill societal needs (ends) by means

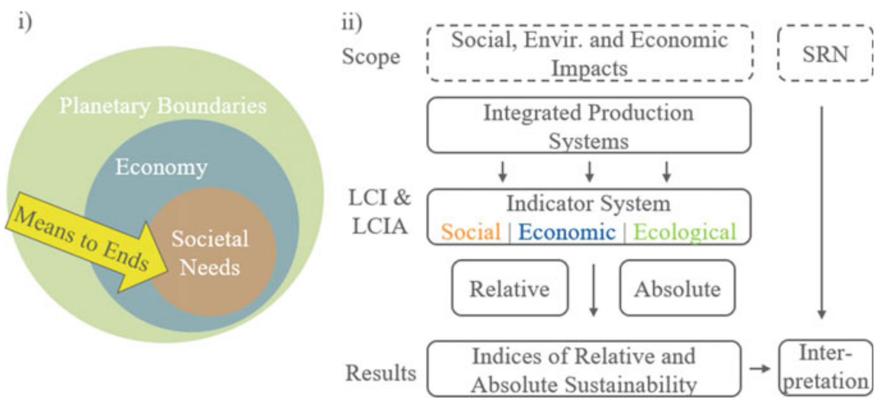


Fig. 8.4 (i) Sustainability model, (ii) Framework of HILCSA = f (S-LCA, E-LCA, LCC) (integrated product and production systems in openLCA entail ecological, social and economic data)

of natural resources, labor and technologies (means). This leads to a model (Fig. 8.4i) in which integrated sustainability is defined as:

- Long-term and global fulfillment of societal needs and well-being as an end (social sustainability)
- Long-term stability of our environment as a basis of societal reproduction within PB (ecological sustainability)
- Technologies and economic structures as efficient, effective, sufficient and just metabolisms which enable the fulfillment of societal needs within PB (economic sustainability)

Economic sustainability in this sense is the enabling criteria for actually reaching social sustainability and ecological sustainability at once, profit or growth is neither a criterion nor an end itself. In a phase before or at the beginning of a societal-ecological transformation, economic sustainability means at least to fulfill most societal needs with the lowest resource use possible.

Between indicators or sustainability aspects there is no compensation or credit (e.g. positive assessment results of indicators are offset with negative results of other indicators in indices) applied, as it is sometimes suggested in LCSA. Simply because there is no meaningful mechanism of compensating GHG emissions by improvements in health at working conditions within a production system. As well as, not transgressing one PB revokes the transgression of another PB; if only one PB is transgressed a long-term reproduction of societies is at stake.

For allocation and weighting of indicators in HILCSA, certain SDGs can be assigned to societal needs, economy and PB, however, a clean analytical distinction is not possible due to the complex interactions (de Schutter et al. 2019): societal needs (SDG 1, 2, 3, 4, 5, 11, (16, 17)); economy (SDG 6, 7, 8, 9, 10, 12); PB (SDG 13, 14, 15) (Zeug et al. 2019, 2021a, 2020, 2022a). We built a SDG framework in previous studies (Zeug et al. 2019) as well as developed (Zeug et al. 2021a, 2020) and applied (Zeug et al. 2022a, 2023) HILCSA. The SDGs are applicable as a commonly agreed on goal and indicator framework. In the following, we are deepening the discourse for further development and applications of (HI)LCSA approaches.

8.3.1 Operationalization and First Results of HILCSA Case Study on Laminated Veneer Lumber

The common goal and scope of HILCSA is the assessment of social, environmental and economic risks, chances, synergies, trade-offs and contradictions of production systems with a focus on BE (Fig. 8.4ii). Although HILCSA is applicable for production systems in general, the focus on BE is given by specific indicators on i.e. land-use-change, biomass extraction or cumulated energy demands without the net calorific value of biomass for material use. For the LCI, the operational core of

HILCSA are integrated production systems and processes entailing social, ecological and economic data which are modeled in the software environment of openLCA, mainly using the SoCa database (Eisfeldt 2017; Di Noi et al. 2018) completed by additional data gathering (e.g. questionnaires (Jarosch et al. 2020)). The SoCa add-on as a combination of Ecoinvent and PSILCA (Product Social Impact Life Cycle Assessment) database as well as a basic LCSA functionality in openLCA is fundamental to this. The LCI in HILCSA entails a set of 109 quantitative and qualitative indicators for HILCSA capable to address societal needs by 21 indicators, economy by 59 indicators and the PB by 29 indicators (Zeug et al. 2021a). Thereby HILCSA is capable of addressing 15 out of 17 SDGs (SDG 10 & 17 missing yet). For the variety of indicators, we combined several established LCIA methods like ReCiPe, Impact World+, EF 3.0, RESPONSA and SoCa. Assessment of indicator values is based on a progressive regulation of SRN and a societal-ecological transformation, e.g. high efficiency and effectiveness, or less working time and a higher average renumeration lead to better assessment scores.

In a first and previous case study (Zeug et al. 2022a) of substituting steel beams with LVL beams (laminated veneer lumber), for each indicator i which is assigned to a specific subgoal SDG $sSDG$, in openLCA we calculate values x for each process of the LVL product system x^{LVL}_{sSDG}, as well as cumulated (total) values for the whole product system of LVL $x^{LVL}_{sSDG,T}$ and the steel beam $x^{SB}_{sSDG,T}$. All cumulated results of all indicators of our BE product system we finally compare to the product which can be substituted (steel beam), to assess their relative rather than absolute impact. Therefore, we calculate a factor f^{sSDG} called substitution-factor of impact of an indicator (Eq. 8.1), expressing the magnitude of relative sustainability. As aggregation on the SDG level, we calculated weighted mean factors for substitution of impact for each SDG f^{SDG} (Eq. 8.2). As weighting factors, we used the relevances R^{sSDG} of each of the SDG-subgoals in the context of the German BE-monitoring (Zeug et al. 2019). Analogical as well a total substitution-factor of impacts f is calculated on the level of all SDGs (Eq. 8.3).

$$f^{sSDG} = \frac{x^{LVL}_{sSDG,T}}{x^{SB}_{sSDG,T}} \tag{8.1}$$

$$f^{SDG} = \frac{\sum_{sSDG} R^{sSDG} f^{sSDG}}{\sum_{sSDG} R^{sSDG}} \tag{8.2}$$

$$f = \frac{\sum_{SDG} R^{SDG} f^{sSDG}}{\sum_{SDG} R^{SDG}} \tag{8.3}$$

According to the assignment of SDGs to societal needs (SDG 1, 2, 3, 4, 5, 11, (16, 17)), economy (SDG 6, 7, 8, 9, 10, 12) and ecology (SDG 13, 14, 15) we calculated substitution factors of impact for social $f_{social} = 0.31$, ecological $f_{ecological} = 1.01$ and economic $f_{economic} = 0.60$ sustainability. LVL seems to have a way better social sustainability, by having a detailed look at the indicator data and inventory, this is mainly due to the less toxicity of materials, immissions on humans and their working environments, but also higher expenditures for social security and education as well as a lower gender wage gap. However, regional analyzes show that the different technical production processes are not the main cause, but the far more global distribution of primary production chains of the steel industry and thereby externalization of social deprivations are worse (Zeug et al. 2022a) (cf. Backhouse et al. 2021). Such effects get visible by integrated and holistic methodologies including political economy, and would probably be neglected or falsely allocated to technologies in conventional LCA. Additionally, from a quantitative analysis, we see that the most significant negative impacts of LVL production come from forestry and its effects on land use with a substitution factor $f = 18.15$, e.g. LVL production takes up more than 18 times the land use of steel since steel as a fossil resource was accumulated inside the earth whereas wood has to steadily grow on its surface. However, the potential impact on climate change due to land use change in total is better than that of steel $f = 0.96$ as well as the overall potential negative effects on climate change are far less $f = 0.39$.

In a nutshell, although BE in this case can substitute fossil materials and partly has lower negative impacts (relative decoupling), forestry and agriculture use relatively much more land for primary resource production than fossil resources (Bringezu et al. 2020; O'Brien et al. 2017; Liobikiene et al. 2020). If BE is only seen as a substitution of resources in a capitalist and growing economy, then PB like land use will be transgressed way faster than in a fossil economy. In other words, substituting fossil resources with renewable resources under the same quantitative and qualitative production and consumption patterns will be unsustainable and makes an absolute decoupling seem implausible. Achieving ultimately more sustainability seems to be very unlikely by bioeconomy alone, but when bioeconomy is embedded in a societal-ecological transformation. Processes based on renewable resources in specific regions do not only have a better ecological, but also better social and economic sustainability as synergies. However, the dependency on sustainability from regions does not only apply to fossil industries, but BE can be very unsustainable when renewable material flows reproduce global social and economic inequalities and externalization of effects of sourcing and production (Asada et al. 2020; Backhouse et al. 2021; Eversberg and Holz 2020).

8.3.2 Further Development of HILCSA and LCA

SRN and a societal-ecological transformation as societal and a techno-economical decoupling have far reaching implications on HILCSA and LCA in general, significant for their further development, e.g. identifying seemingly technological problems as embedded problems of political economy and addressing them from a critical and transdisciplinary perspective. Currently, social sustainability in LCA and HILCSA is only measured as potential direct and indirect impacts of production on health, well-fare, education, (gender) equality, etc. of workers and communities in general.

Regarding a techno-economical decoupling, HILCSA currently aims to create an overview of the sustainability of production systems, as complete and concrete as possible. Risks, chances, synergies, trade-offs and hot spots are identified, whereas trade-offs, in particular, are important since they indicate contradictions which are characteristic of capitalists' patterns of regulation and metabolisms and should be avoided in a societal-ecological transformation. As outlined before, surplus and exchange values dominate use value and consequentially monetarization of social, ecological and economic aspects impacts LCA and LCSAs as well. A problem of fundamental character appears, which has not been discussed extensively in the previous research yet: to what extent monetary variables are generally distorted and abstract representations of (non–)material objects, subjects and their relationships in form of exchange values. In contrast to physical quantities, costs and prices are subject to abstract quantities and substantial fluctuations, not only due to fluctuations in market prices due to changing (un-)equilibria of supply and demand. For example, the amount of CO_2 emitted when a certain amount of a fuel is burned and the subsequent effects on the atmosphere and climate change are almost independent of location and, in the short term, time. Most internalized costs, on the other hand, for one and the same commodity can depend both in real and nominal terms on several factors, such as region, currency and time, and show significant differences (Ciroth 2009). As well as accounting procedures themselves are not standardized (Swarr et al. 2011). Besides, solely costs are of secondary importance for the production and marketing of commodities under capitalism; the prospect of a return on capital and profit remains paramount (Ciroth 2009; Postone 1993; Zeug et al. 2020). As well as decisive for most economic decisions are not the absolute balanced costs, but the relative costs of the opportunities (Kuosmanen 2005). For this series of reasons as well as potential future applications (Sect. 8.4), HILCSA avoids monetarization and relies primarily on material and energy flows as well as working time for balancing. Indicators representing economic sustainability are i.e. water and energy consumption, share of fossil energies, resource efficiency, cascading factor, weekly hours of work per employee, average remuneration level, children in employment, and right of association (Zeug et al. 2021a). In addition, life cycle costs are also implemented as a variable.

A challenge will be that private industrial actors in capitalist societal relations have and must have an intrinsic interest in capital accumulation and increasing output, and by themselves will not embark on a good life for all within PB or cost internalization. Societal decoupling will in particular rely on a decreasing production of material goods and is essentially coherent with techno-economic decoupling not transgressing PB by resource use and environmental impacts is a hard criterion. Consequentially, beyond the importance of regionalized and spatially explicit datasets in order to improve the quality of results (Chandrakumar and McLaren 2018a, b; Chandrakumar et al. 2018). In recent years, significant developments were made, especially in the context of the European Commission—Joint Research Centre (EC-JRC) to integrate PB and environmental footprints (EF) into E-LCA to allow meso- and macroeconomic assessments and conclusions by sector and product specific bottom-up approaches (Bjørn et al. 2020; Robert et al. 2020; Sala and Castellani 2019; Sala et al. 2020). Like a majority of LCAs, HILCSA as well entails a relative assessment, e.g. if the observed case is better than a reference of cases and how much it is (substitution factor of impacts). However, there is no information on if it performs 'well enough' for ecological sustainability in terms of PB (Bjørn et al. 2020). Whereas absolute sustainability assessment methods (Chandrakumar and McLaren 2018b) compare specific impacts with external environmental carrying capacities (according to PB), e.g. life-cycle climate impacts are related to the 1.5° climate goal (Bjørn et al. 2020). In a relative dimension, this comes down to assessing how much kg CO_2 eq. per product can be considered as (un-)sustainable, however, on an absolute dimension it is a question of what quantities of such a product can be produced in general within a specific time frame. Such PB-LCIAs (Ryberg et al. 2018) addressing challenges of relating LCIs and LCIAs to operational definitions of PBs (Robert et al. 2020) are significant for BE, since a sustainable BE requires that the rate of extraction does not exceed the rate of regeneration, and that this regenerativity and the surrounding supporting systems are maintained. However, such absolute sustainability assessment methods are not robustly available in LCA, yet (Alejandrino et al. 2021; Guinée et al. 2022). The major reason and hurdle, besides technical complexity, are so-called problems of sharing principles and distributive justice theories used in diverse political philosophies (i.e. egalitarian, utilitarian, and acquired rights principles) Ryberg et al. 2020, 2018), e.g. the basic question to determine how much products and resources of whole economies can be granted to different social entities (individuals, regions, nations). We consider addressing these questions requires societal and democratic political processes as well as a transdisciplinary scientific perspective for which HILCSA can provide a specific tool, data, information and interpretations.

8.4 Conclusions and Outlook

At this very point, the mutual dependency and relation of societal decoupling and techno-economic decoupling (PB) leads unavoidably to fundamental questions of political economy and political ecology: How to socially organize and normatively analyze the fulfillment of societal needs by economies within PBs? For various previously mentioned reasons, but especially due to the twisted relations of means and ends, this question is unlikely to be solved within capitalists' societal relations and their intrinsic compulsion to grow, independent of which 'philosophy' is applied. On the other hand, in political economy and ecology, a new discourse is rising in the direction of which the approaches of an absolute sustainability assessment and HILCSA point implicitly: new forms of distributed planned economies. Planning economy means to mentally, organizationally and institutionally shape processes of determining, through assessment and decisions, on which paths, with which steps, in which temporal and organizational sequence, under which framework conditions and finally with which 'costs' and consequences a certain goal seems to be achievable

(Nuss and Daum 2021). Of course, planning in this regard, as a mental anticipation of actions, is already immanent for the current economic system, especially in times of large digital platforms but under very different preconditions (Bastani 2019; Morozov 2019; Phillips and Rozworski 2019). Climate change as a relatively new global problem can only be countered by means of collective planning, however, the debate on capitalist market economies versus socialist planned economies has a long tradition and comes down to the question of which societal and technical basis, how and supported by which tools an economy is organized and coordinated (Groos 2021). Against the background of societal decoupling, it would be of particular interest to implement whether and to what extent the product manufactured and evaluated actually meets social needs in terms of effectiveness, sufficiency and justice.

For such future theoretical perspectives as well as current assessment, HILCSA allows an integrative and holistic sustainability analysis and assessment based on aggregated indicators qualitative discussion, retrospective and prospective. At this early stage, the indicator and impact assessment sets are not as detailed as in the stand alone methods, rather the goal is to avoid a piecemeal approach to SD (Taylor et al. 2017) and to deliver a comprehensive picture of trade-offs, synergies, hotspots, significant risks and chances and a fundamental understanding. Currently, the techno-economic dimension of decoupling can be described relatively well, the societal dimension of decoupling only partially with the need for transdisciplinary cooperation and integration. At this point, however, LCSAs can no longer be sharply and meaningfully separated from political and macroeconomic topics, which was proposed in additive LCSA. For further applications in regional production systems and macroeconomic systems, the extension towards multi regional input output methods (MRIO) and hybrid LCSAs is promising (Budzinski et al. 2017; Fröhlich 2009; Jander and Grundmann 2019; Teh et al. 2017).

BE and circular economy as well as sustainability assessments are for both societal-ecological transformations and "green" capitalism necessary and meaningful. Less unsustainable practices even under SRN of capitalism are viable to retain the environmental basis for anything beyond. However, the overall possibilities of achieving sustainability by BE and sustainability assessments are limited as long as social, ecological and economic sustainability are not a central objective of the general economy and its patterns of regulation itself.

References

Aguilar A, Wohlgemuth R, Twardowski T (2018) Preface to the special issue bioeconomy. New Biotechnol 40:1–4. https://doi.org/10.1016/j.nbt.2017.06.008

Alejandrino C, Mercante I, Bovea MD (2021) Life cycle sustainability assessment: lessons learned from case studies. Environ Impact Assess Rev 87 106517ARTN106517. https://doi.org/10.1016/j.eiar.2020.106517

Amidon KS (2008) Diesmal fehlt die Biologie! Max Horkheimer, Richard Thurnwald, and the Biological Prehistory of German Sozialforschung. New German Critique 103–137

Asada R, Cardellini G, Mair-Bauernfeind C, Wenger J, Haas V, Holzer D, Stern T (2020) Effective bioeconomy? a MRIO-based socioeconomic and environmental impact assessment of generic sectoral innovations. Technol Forecast Soc Change 153:119946. https://doi.org/10.1016/j.techfore.2020.119946

Ashukem J-CN (2020) The SDGs and the bio-economy: fostering land-grabbing in Africa. Rev Afr Polit Econ 1–16. https://doi.org/10.1080/03056244.2019.1687086

Backhouse M, Lehmann R, Lorenzen K, Lühmann M, Puder J, Rodríguez F, Tittor A (2021) Bioeconomy and global inequalities. Springer Nature. https://doi.org/10.1007/978-3-030-68944-5

Barbier EB (1999) Endogenous growth and natural resource scarcity. Environ Resource Econ 14:51–74. https://doi.org/10.1023/a:1008389422019

Bastani A (2019) Fully automated luxury communism. Verso Books

Bauriedl S (2016) Politische Ökologie: nicht-deterministische, globale und materielle Dimensionen von Natur/Gesellschaft-Verhältnissen Geographica Helvetica 71:341–351. https://doi.org/10.5194/gh-71-341-2016

Bell J, Paula L, Dodd T, Németh S, Nanou C, Mega V, Campos P (2018) EU ambition to build the world's leading bioeconomy—uncertain times demand innovative and sustainable solutions. New Biotechnol 40:25–30. https://doi.org/10.1016/j.nbt.2017.06.010

Bennich T, Belyazid S (2017) The route to sustainability—prospects and challenges of the bio-based economy. Sustainability 9:887

Bezama A, Mittelstädt N, Thrän D (2021) A systematic approach for assessing and managing the urban bioeconomy. In: Koukios E, Sacio-Szymańska A (eds) Bio#futures: foreseeing and exploring the bioeconomy. Springer International Publishing, Cham, pp 393–410. https://doi.org/10.1007/978-3-030-64969-2_18

Bioökonomierat (2022) Bioökonomie: Gemeinsam eine nachhaltige Zukunft gestalten - 1. Arbeitspapier des III. Bioökonomierats, Berlin

Birner R (2018) Bioeconomy Concepts. In: Lewandowski I (ed) Bioeconomy: shaping the transition to a sustainable, biobased economy. Springer International Publishing, Cham, pp 17–38. doi:https://doi.org/10.1007/978-3-319-68152-8_3

Bjørn A et al (2020) Review of life-cycle based methods for absolute environmental sustainability assessment and their applications. Environ Res Lett 15:083001. https://doi.org/10.1088/1748-9326/ab89d7

Brand U, Wissen M (2018) The limits to capitalist nature: theorizing and overcoming the imperial mode of living. Rowman & Littlefield International

Bringezu S et al. (2020) Pilotbericht zum Monitoring der deutschen Bioökonomie. Center for Environmental Systems Research (CESR), Kassel. https://doi.org/10.17170/kobra-202005131255

Brundtland G et al (1987) Our common future ('Brundtland report'). Oxford University Press, USA

Brynjolfsson E, Andrew M (2015) The great decoupling

Budzinski M, Bezama A, Thran D (2017) Monitoring the progress towards bioeconomy using multi-regional input-output analysis: The example of wood use in Germany. J Clean Prod 161:1–11. https://doi.org/10.1016/j.jclepro.2017.05.090

Chandrakumar C, McLaren SJ (2018a) Exploring the linkages between the environmental sustainable development goals and planetary boundaries using the dpsir impact pathway framework. In: Benetto E, Gericke K, Guiton M (eds) Designing sustainable technologies, products and policies. Springer International Publishing, Cham, pp 413–423. https://doi.org/10.1007/978-3-319-66981-6_46

Chandrakumar C, McLaren SJ (2018b) Towards a comprehensive absolute sustainability assessment method for effective Earth system governance: defining key environmental indicators using an enhanced-DPSIR framework. Ecol Ind 90:577–583. https://doi.org/10.1016/j.ecolind.2018.03.063

Chandrakumar C, McLaren SJ, Jayamaha NP, Ramilan T (2018) Absolute Sustainability-Based Life Cycle Assessment (ASLCA): a benchmarking approach to operate agri-food systems within the 2 °C global carbon budget. J Ind Ecol 23:906–917. https://doi.org/10.1111/jiec.12830

Ciroth A (2009) Cost data quality considerations for eco-efficiency measures. Ecol Econ 68:1583–1590. https://doi.org/10.1016/j.ecolecon.2008.08.005

Common M, Stagl S (2012) Ecological economics. Cambridge University Press, Cambridge. https://doi.org/10.1017/cbo9780511805547

Costa D, Quinteiro P, Dias AC (2019) A systematic review of life cycle sustainability assessment: current state, methodological challenges, and implementation issues. Sci Total Environ 686:774–787. https://doi.org/10.1016/j.scitotenv.2019.05.435

Costanza R, Cumberland J, Daly H, Goodland R, Norgaard R (1997) An introduction to ecological economics. CRC Press

D'Amato D, Gaio M, Semenzin E (2020) A review of LCA assessments of forest-based bioeconomy products and processes under an ecosystem services perspective. Sci Total Environ 706:135859. https://doi.org/10.1016/j.scitotenv.2019.135859

D'Amato D (2021) Sustainability narratives as transformative solution pathways: zooming in on the circular economy. Circ Econ Sustain 1:231–242. https://doi.org/10.1007/s43615-021-00008-1

de Besi M, McCormick K (2015) Towards a bioeconomy in Europe: national regional and industrial strategies. Sustainability 7:10461–10478. https://doi.org/10.3390/su70810461

de Schutter L, Giljum S, Hayha T, Bruckner M, Naqvi A, Omann I, Stagl S (2019) Bioeconomy transitions through the lens of coupled social-ecological systems: a framework for place-based responsibility in the global resource system. Sustainability 11:5705. https://doi.org/10.3390/su11205705

Demirovic A (2003) Vorwort. In: Demirovic A (ed) Modelle kritischer Gesellschaftstheorie. J.B. Metzler, Stuttgart, pp 1–9

Di Noi C, Eisfeldt F, Ciroth A, Bizarro D (2018) Complementarity of social and environmental indicators and risks. An example of the mining industry. Paper presented at the S-LCA 2018, Pescara, September 2018

Egenolf V, Bringezu S (2019) Conceptualization of an indicator system for assessing the sustainability of the bioeconomy. Sustainability 11:443

Eisfeldt F (2017) Soca v.1 add-on—adding social impact information to ecoinvent. GreenDelta GmbH, Berlin

Ekener E, Hansson J, Larsson A, Peck P (2018) Developing life cycle sustainability assessment methodology by applying values-based sustainability weighting—tested on biomass based and fossil transportation fuels. J Clean Prod 181:337–351. https://doi.org/10.1016/j.jclepro.2018.01.211

Elkington J (1998) Partnerships fromcannibals with forks: the triple bottom line of 21st-century business. Environ Qual Manage 8:37–51. https://doi.org/10.1002/tqem.3310080106

Erb K-H, Haberl H, Le Noe J, Tappeiner U, Tasser E, Gingrich S (2022) Changes in perspective needed to forge "no-regret" forest-based climate change mitigation strategies. GCB Bioenergy 14. https://doi.org/10.1111/gcbb.12921

Eversberg D, Fritz M (2022) Bioeconomy as a societal transformation: mentalities, conflicts and social practices. Sustain Prod Consum 30:973–987. https://doi.org/10.1016/j.spc.2022.01.021

Eversberg D, Holz J (2020) Empty promises of growth: the bioeconomy and its multiple reality checks. In: Working paper #2 of the BMBF Junior Research Group "Mentalitites in Flux" (flumen). Friedrich-Schiller-Universität Jena, Jena. https://doi.org/10.13140/RG.2.2.30275.84007

Fauzi RT, Lavoie P, Sorelli L, Heidari MD, Amor B (2019) Exploring the current challenges and opportunities of life cycle sustainability assessment. Sustainability 11:636. https://doi.org/10.3390/su11030636

Fedrigo-Fazio D, Schweitzer J-P, Ten Brink P, Mazza L, Ratliff A, Watkins E (2016) Evidence of absolute decoupling from real world policy mixes in Europe. Sustainability 8. https://doi.org/10.3390/su8060517

Fröhlich N (2009) Die Aktualität der Arbeitswerttheorie - Theoretische und empirische Aspekte vol 128. Hochschulschriften. Metropolis, Marburg
Future Earth (2016) The contribution of science in implementing the sustainable development goals. German Committee Future Earth, Stuttgart
Gawel E, Pannicke N, Hagemann N (2019) A path transition towards a bioeconomy—the crucial role of sustainability. Sustainability 11:3005. https://doi.org/10.3390/su11113005
Georgescu-Roegen N (1971) The entropy law and the economic process
Gerhardt P (2018) Bioökonomie – die neue Nebelwand aus der PR-Abteilung. denkhaus-bremen. https://denkhausbremen.de/biooekonomie-die-neue-nebelwand-aus-der-pr-abteilung/. Accessed 19 Novemvber 2018
German Bioeconomy Council (2018a) http://biooekonomierat.de/en/
German Bioeconomy Council (2018b) Update report of national strategies around the world—bioeconomy policy (Part III). Bioeconomy Council, Berlin
GFN GFN (2019) Ecological deficit/reserve. https://data.footprintnetwork.org. Accessed 05 September 2019
Görg C (2003) Dialektische Konstellationen. Zu einer kritischen Theorie gesellschaftlicher Naturverhältnisse. In: Demirovic A (ed) Modelle kritischer Gesellschaftstheorie. J.B. Metzler, Stuttgart, pp 39–62
Görg C (2004) The construction of societal relationships with nature. Poiesis & Praxis 3:22–36. https://doi.org/10.1007/s10202-004-0066-5
Görg C (2011) Societal Relationships with nature: a dialectical approach to environmental politics. In: Biro A (ed) Critical ecologies. University of Toronto Press, Toronto
Görg C (2015) Planetarische Grenzen. In: Bauriedl S (ed) Wörterbuch Klimadebatte, vol 82. transcript Verlag
Görg C, Brand U, Haberl H, Hummel D, Jahn T, Liehr S (2017) Challenges for social-ecological transformations: contributions from social and political ecology. Sustainability 9:1045
Groos J (2021) Distributed planned economies in the age of their technical feasibility. BEHEMOTH A J Civil 14. https://doi.org/10.6094/behemoth.2021.14.2.1061
Guinée J (2016) life cycle sustainability assessment: what is it and what are its challenges? In: Clift R, Druckman A (eds) Taking stock of industrial ecology. Springer International Publishing, Cham, pp 45–68. https://doi.org/10.1007/978-3-319-20571-7_3
Guinee JB et al (2011) Life cycle assessment: past, present, and future. Environ Sci Technol 45:90–96. https://doi.org/10.1021/es101316v
Guinée JB et al (2011) Life cycle assessment: past, present, and future. Environ Sci Technol 45:90–96. https://doi.org/10.1021/es101316v
Guinée JB, Koning A, Heijungs R (2022). Life cycle assessment-based absolute environmental sustainability assessment is also relative. J Ind Ecol n/a. https://doi.org/10.1111/jiec.13260
Haberl H, Steinberger JK, Plutzar C, Erb K-H, Gaube V, Gingrich S, Krausmann F (2012) Natural and socioeconomic determinants of the embodied human appropriation of net primary production and its relation to other resource use indicators. Ecol Ind 23:222–231. https://doi.org/10.1016/j.ecolind.2012.03.027
Halog A, Manik Y (2011) Advancing integrated systems modelling framework for life cycle sustainability assessment
Harvey D, Braun B (1996) Justice, nature and the geography of difference, vol 468. Blackwell Oxford
Hauff M, Jörg A (2013) Nachhaltiges Wachstum. Oldenbourg Verlag, München
Hausknost D, Schriefl E, Lauk C, Kalt G (2017) A transition to which bioeconomy? an exploration of diverging techno-political choices. Sustainability 9:669. https://doi.org/10.3390/su9040669
Hector D, Christensen C, Petrie J (2014) Sustainability and sustainable development: philosophical distinctions and practical implications, vol 23. https://doi.org/10.3197/096327114X13851122268963
Holmberg J, Environment IIf, Development (1992) Policies for a small planet: from the international institute for environment and development. Earthscan

Hopwood B, Mellor M, O'Brien G (2005) Sustainable development: mapping different approaches. Sustain Dev 13:38–52. https://doi.org/10.1002/sd.244
Hummel D, Jahn T, Keil F, Liehr S, Stieß I (2017) Social ecology as critical, transdisciplinary science—conceptualizing. analyzing and shaping societal relations to nature. Sustainability 9:1050
Ingrao C et al. (2018) The potential roles of bio-economy in the transition to equitable, sustainable, post fossil-carbon societies: findings from this virtual special issue. J Clean Prod 204:471–488. https://doi.org/10.1016/j.jclepro.2018.09.068
IPCC (2022) Climate change 2022—impacts adaption and vulnerability. UNEP
Jahn T, Bergmann M, Keil F (2012) Transdisciplinarity: between mainstreaming and marginalization. Ecol Econ 79:1–10. https://doi.org/10.1016/j.ecolecon.2012.04.017
Jander W, Grundmann P (2019) Monitoring the transition towards a bioeconomy: a general framework and a specific indicator. J Clean Prod 236:117564. https://doi.org/10.1016/j.jclepro.2019.07.039
Jarosch L, Zeug W, Bezama A, Finkbeiner M, Thran D (2020) A regional socio-economic life cycle assessment of a bioeconomy value chain. Sustainability 12. https://doi.org/10.3390/su12031259
Karabarbounis L, Neiman B (2013) The global decline of the labor share. National Bureau of Economic Research Working Paper Series No. 19136. https://doi.org/10.3386/w19136
Keller H, Rettenmaier N, Reinhardt GA (2015) Integrated life cycle sustainability assessment—a practical approach applied to biorefineries. Appl Energy 154:1072–1081. https://doi.org/10.1016/j.apenergy.2015.01.095
Klein JT (2008) Evaluation of interdisciplinary and transdisciplinary research: a literature review. Am J Prev Med 35:S116-123. https://doi.org/10.1016/j.amepre.2008.05.010
Kleinschmit D, Arts B, Giurca A, Mustalahti I, Sergent A, Pulzl H (2017) Environmental concerns in political bioeconomy discourses. Int Forest Rev 19:41–55. https://doi.org/10.1505/146554817822407420
Kloepffer W (2008) Life cycle sustainability assessment of products. Int J Life Cycle Assess 13:89–95. https://doi.org/10.1065/lca2008.02.376
Knierim A, Laschewski L, Boyarintseva O (2018) Inter- and transdisciplinarity in bioeconomy. In: Lewandowski I (ed) Bioeconomy: shaping the transition to a sustainable, biobased economy. Springer International Publishing, Cham, pp 39–72. https://doi.org/10.1007/978-3-319-68152-8_4
Kramm J, Pichler M, Schaffartzik A, Zimmermann M (2017) Societal relations to nature in times of crisis—social ecology's contributions to interdisciplinary sustainability studies. Sustainability 9:1042
Kuosmanen T (2005) Measurement and analysis of eco-efficiency—an economist's perspective. J Ind Ecol 9:15–18. https://doi.org/10.1162/108819805775248025
Lamont M, Mallard G, Guetzkow J (2006) Beyond blind faith: overcoming the obstacles to interdisciplinary evaluation. Res Evaluat 15:43–55. https://doi.org/10.3152/147154406781776002
Lindqvist AN, Broberg S, Tufvesson L, Khalil S, Prade T (2019) Bio-based production systems: why environmental assessment needs to include supporting systems. Sustainability 11:4678. https://doi.org/10.3390/su11174678
Liu J et al (2015) Systems integration for global sustainability. Science 347:1258832. https://doi.org/10.1126/science.1258832
Lubchenco J, Barner AK, Cerny-Chipman EB, Reimer JN (2015) Sustain Rooted Sci Nat Geosci 8:741. https://doi.org/10.1038/ngeo2552
Meadowcroft J (2007) Who is in charge here? governance for sustainable development in a complex world. J Environ Policy Plan 9:299–314. https://doi.org/10.1080/15239080701631544
Mebratu D (1996) Sustainability as a scientific paradigm. Lund: Int Inst Ind Environ Econ
Mebratu D (1998) Sustainability and sustainable development: historical and conceptual review. Environ Impact Assess Rev 18:493–520. https://doi.org/10.1016/S0195-9255(98)00019-5

Menton M, Larrea C, Latorre S, Martinez-Alier J, Peck M, Temper L, Walter M (2020) Environmental justice and the SDGs: from synergies to gaps and contradictions. Sustain Sci. https://doi.org/10.1007/s11625-020-00789-8
Meyer R (2017) Bioeconomy Strategies: Contexts. Visions, guiding implementation principles and resulting debates. Sustainability 9:1031
Michel P, Rotillon G (1995) Disutility of pollution and endogenous growth. Environ Resour Econ 6:279–300. https://doi.org/10.1007/bf00705982
Morozov E (2019) Digital socialism? The calculation debate in the age of big data. New Left Rev 33–67
Mustalahti I (2018) The responsive bioeconomy: the need for inclusion of citizens and environmental capability in the forest based bioeconomy. J Clean Prod 172:3781–3790. https://doi.org/10.1016/j.jclepro.2017.06.132
Nerini F et al (2019) Connecting climate action with other sustainable development Goals. Nat Sustain 2:674–680. https://doi.org/10.1038/s41893-019-0334-y
Network GF (2019) Global footprint network glossary. https://www.footprintnetwork.org/resources/glossary/. Accessed 27 November 2019
Nitzsche R, Budzinski M, Gröngröft A (2016) Techno-economic assessment of a wood-based biorefinery concept for the production of polymer-grade ethylene, organosolv lignin and fuel. Bioresour Technol 200:928–939. https://doi.org/10.1016/j.biortech.2015.11.008
Norström AV (2013) Social change vital to sustainability goals. Nature 498:299. https://doi.org/10.1038/498299c. https://www.nature.com/articles/498299c#supplementary-information
Nuss S, Daum T (2021) The invisible hand of the plan—coordination and calculation in digital capitalism [Die unsichtbare Hand des Plans - Koordination und Kalkül im digitalen Kapitalismus]. Dietz, Berlin
O'Neill DW, Fanning AL, Lamb WF, Steinberger JK (2018) A good life for all within planetary boundaries. Nat Sustain 1:88–95. https://doi.org/10.1038/s41893-018-0021-4
OECD (2018) Meeting policy challenges for a sustainable bioeconomy. OECD Publishing, Paris. https://doi.org/10.1787/9789264292345-en
Olsson O et al. (2016) Cascading of woody biomass: definitions, policies and effects on international trade. IEA Bioenergy
Palahí M (2021) Is forest harvesting increasing in Europe? European Forest Institute. https://efi.int/articles/nature. Accessed 15 June 2022
Panoutsou C, Manfredi S, Kavalov B (2013) Biomass resource efficiency for the biobased industries. Joint Research Centre (JRC), Ispra
Parrique TBJ, Briens F, Kerschner C, Kraus-Polk A, Kuokkanen A, Spangenberg JH (2019) Decoupling Debunked - Evidence and arguments against green growth as a sole strategy for sustainability. The European Environmental Bureau, Brussels
Perdomo Echenique EA, Ryberg M, Vea EB, Schwarzbauer P, Hesser F (2022) Analyzing the consequences of sharing principles on different economies: a case study of short rotation coppice poplar wood panel production value chain. Forests 13:461
Pfau SF, Hagens JE, Dankbaar B, Smits AJM (2014) Visions of sustainability in bioeconomy research. Sustainability 6:1222–1249. https://doi.org/10.3390/su6031222
Phillips L, Rozworski M (2019) The people's republic of Walmart: how the world's biggest corporations are laying the foundation for socialism. Verso Books
Pichler M, Brand U, Görg C (2020) The double materiality of democracy in capitalist societies: challenges for social-ecological transformations. Env Polit 29:193–213. https://doi.org/10.1080/09644016.2018.1547260
Pichler M, Schaffartzik A, Haberl H, Görg C (2017) Drivers of society-nature relations in the Anthropocene and their implications for sustainability transformations. Curr Opin Environ Sustain 26–27:32–36. https://doi.org/10.1016/j.cosust.2017.01.017
Piketty T (2014) Capital in the twenty-first century. Harvard University Press
Postone M (1993) Time, labor, and social domination. Cambridge University Press, New York

Purvis B, Mao Y, Robinson D (2019) Three pillars of sustainability: in search of conceptual origins. Sustain Sci 14:681–695. https://doi.org/10.1007/s11625-018-0627-5
Ramcilovic S, Pülzl H (2018) Sustainable development—a 'selling point' of the emerging EU bioeconomy policy framework? J Clean Prod 172:4170–4180. https://doi.org/10.1016/j.jclepro.2016.12.157
Redclift MR, Benton T (1994) Social theory and the global environment. Routledge
Robert N et al (2020) Development of a bioeconomy monitoring framework for the European Union: an integrative and collaborative approach. N Biotechnol 59:10–19. https://doi.org/10.1016/j.nbt.2020.06.001
Rockström J et al (2009) A Safe operating space for humanity nature 461:472. https://doi.org/10.1038/461472a
Rockström J, Sukhdev P (2016) How food connects all the SDGs. Stockholm Resilience Centre. http://www.stockholmresilience.org/research/research-news/2016-06-14-how-food-connects-all-the-sdgs.html. Accessed 01 November 2017
Rosenfield PL (1992) The potential of transdisciplinary research for sustaining and extending linkages between the health and social sciences. Soc Sci Med 35:1343–1357. https://doi.org/10.1016/0277-9536(92)90038-R
Roser M (2022) Our world in data, published online at OurWorldInData.org. Global Change Data Lab. https://ourworldindata.org/
Ryberg MW, Andersen MM, Owsianiak M, Hauschild MZ (2020) Downscaling the planetary boundaries in absolute environmental sustainability assessments—a review. J Clean Prod 276:123287. https://doi.org/10.1016/j.jclepro.2020.123287
Ryberg MW, Owsianiak M, Richardson K, Hauschild MZ (2018) Development of a life-cycle impact assessment methodology linked to the Planetary Boundaries framework. Ecol Ind 88:250–262. https://doi.org/10.1016/j.ecolind.2017.12.065
Sala S, Castellani V (2019) The consumer footprint: monitoring sustainable development goal 12 with process-based life cycle assessment. J Clean Prod 240:118050. https://doi.org/10.1016/j.jclepro.2019.118050
Sala S, Crenna E, Secchi M, Sanye-Mengual E (2020) Environmental sustainability of European production and consumption assessed against planetary boundaries. J Environ Manage 269:110686. https://doi.org/10.1016/j.jenvman.2020.110686
Sala S, Farioli F, Zamagni A (2012a) Life cycle sustainability assessment in the context of sustainability science progress (part 2). Int J Life Cycle Assess 18:1686–1697. https://doi.org/10.1007/s11367-012-0509-5
Sala S, Farioli F, Zamagni A (2012b) Progress in sustainability science: lessons learnt from current methodologies for sustainability assessment: Part 1. Int J Life Cycle Assess 18:1653–1672. https://doi.org/10.1007/s11367-012-0508-6
Schaefer F, Luksch U, Steinbach N, Cabeça J, Hanauer J (2006) Ecological footprint and biocapacity—the world's ability to regenerate resources and absorb waste in a limited time period. European Commission, Luxembourg
Schleyer C, Lux A, Mehring M, Görg C (2017) Ecosystem services as a boundary concept: arguments from social ecology. Sustainability 9:1107
Šimunović N, Hesser F, Stern T (2018) Frame analysis of ENGO conceptualization of sustainable forest management: environmental justice and neoliberalism at the core of sustainability. Sustainability 10:3165
Smulders S (1995) Entropy, environment, and endogenous economic growth. Int Tax Pub Finance 2:319–340. https://doi.org/10.1007/bf00877504
Staffas L, Gustavsson M, McCormick K (2013) Strategies and policies for the bioeconomy and bio-based economy: an analysis of official national approaches. Sustainability 5:2751–2769. https://doi.org/10.3390/su5062751
Steffen W et al (2018) Trajectories of the earth system in the anthropocene. Proc Natl Acad Sci U S A 115:8252–8259. https://doi.org/10.1073/pnas.1810141115

Stegemann L, Ossewaarde M (2018) A sustainable myth: a neo-Gramscian perspective on the populist and post-truth tendencies of the European green growth discourse. Energy Res Soc. Sci 43:25–32. https://doi.org/10.1016/j.erss.2018.05.015

Stern T, Ploll U, Spies R, Schwarzbauer P, Hesser F, Ranacher L (2018) Understanding perceptions of the bioeconomy in austria—an explorative case study. Sustainability 10:4142

Suwelack K (2016) Conversion technology and life cycle assessment of renewable resources. Hohenheim University, Hohenheim

Swarr TE, Hunkeler D, Klopffer W, Pesonen HL, Ciroth A, Brent AC, Pagan R (2011) Environmental life-cycle costing: a code of practice. Int J Life Cycle Ass 16:389–391. https://doi.org/10.1007/s11367-011-0287-5

Taylor PG, Abdalla K, Quadrelli R, Vera I (2017) Better energy indicators for sustainable development. Nat Energy 2:17117. https://doi.org/10.1038/nenergy.2017.117

Teh SH, Wiedmann T, Schinabeck J, Moore S (2017) Replacement scenarios for construction materials based on economy-wide hybrid LCA. Procedia Eng 180:179–189. https://doi.org/10.1016/j.proeng.2017.04.177

Traverso M, Finkbeiner M, Jorgensen A, Schneider L (2012) Life cycle sustainability dashboard. J Ind Ecol 16:680–688. https://doi.org/10.1111/j.1530-9290.2012.00497.x

Troullaki K, Rozakis S, Kostakis V (2021) Bridging barriers in sustainability research: a review from sustainability science to life cycle sustainability assessment. Ecol Econ 184:107007. https://doi.org/10.1016/j.ecolecon.2021.107007

Trzyna TC, Osborn JK, Nature IUfCo, Resources N (1995) A sustainable world: defining and measuring sustainable development. Published for IUCN—the World Conservation Union by the International Center for the Environment and Public Policy, California Institute of Public Affairs

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

UNEP (2020) Guidelines for social life cycle assessment of products and organizations. United Nations Environment Programme (UNEP)

Urban K et al (2018) Markets, sustainability management and entrepreneurship. In: Lewandowski I (ed) Bioeconomy. Springer International Publishing, Cham, pp 231–286. https://doi.org/10.1007/978-3-319-68152-8_8

Valdivia S et al (2021) Principles for the application of life cycle sustainability assessment. Int J Life Cycle Assess 26. https://doi.org/10.1007/s11367-021-01958-2

Verdier T (1995) Environmental pollution and endogenous growth. In: Boston MA (ed) Control and game-theoretic models of the environment. Birkhäuser Boston, pp 175–200

Victor TYH, Chang P, Blackburn K (1994) Endogenous growth, environment and R&D. In: Carraro C (ed) Trade, innovation, environment. Springer Netherlands, Dordrecht, pp 241–258. https://doi.org/10.1007/978-94-011-0948-2_10

Ward JD, Sutton PC, Werner AD, Costanza R, Mohr SH, Simmons CT (2016) Is decoupling GDP growth from environmental impact possible? PLoS ONE 11:e0164733. https://doi.org/10.1371/journal.pone.0164733

Wittmer H et al (2022) Transformative change for a sustainable management of global commons—biodiversity, forests and the ocean. Recommendations for international cooperation based on a review of global assessment reports and project experience. https://doi.org/10.57699/7s83-7z35

Wulf C, Werker J, Ball C, Zapp P, Kuckshinrichs W (2019) Review of sustainability assessment approaches based on life cycles. Sustainability 11:5717. https://doi.org/10.3390/su11205717

Zeug W, Bezama A, Moesenfechtel U, Jähkel A, Thrän D (2019) Stakeholders' interests and perceptions of bioeconomy monitoring using a sustainable development goal framework. Sustainability 11:1511. https://doi.org/10.3390/su11061511

Zeug W, Bezama A, Thrän D (2020) Towards a holistic and integrated life cycle sustainability assessment of the bioeconomy—background on concepts, visions and measurements, vol 07. Helmholtz-Centre for Environmental Research (UFZ), Leipzig. https://doi.org/10.13140/RG.2.2.16912.02564

Zeug W, Bezama A, Thran D (2021a). A framework for implementing holistic and integrated life cycle sustainability assessment of regional bioeconomy Int J Life Cycle Ass. https://doi.org/10.1007/s11367-021-01983-1
Zeug W, Bezama A, Thran D (2022) Application of holistic and integrated LCSA: case study on laminated veneer lumber production in Central Germany. Int J Life Cycle Ass 27:1352–1375. https://doi.org/10.1007/s11367-022-02098-x
Zeug W, Bezama A, Thrän D (2023) Holistic and integrated life cycle sustainability assessment of prospective second generation biofuel production in Germany Forthcoming
Zeug W, Kluson F, Mittelstädt N, Bezama A, Thrän D (2021b) Results from a stakeholder survey on bioeconomy monitoring and perceptions on bioeconomy in Germany. Helmholtz-Centre for Environmental Research, Leipzig. https://doi.org/10.13140/RG.2.2.35521.28000
Zimek M, Schober A, Mair C, Baumgartner RJ, Stern T, Füllsack M (2019) The third wave of LCA as the "decade of consolidation. Sustainability 11:3283. https://doi.org/10.3390/su11123283
Zörb C, Lewandowski I, Kindervater R, Göttert U, Patzelt D (2018) Biobased resources and value chains. In: Lewandowski I (ed) Bioeconomy: shaping the transition to a sustainable, biobased economy. Springer International Publishing, Cham, pp 75–95. https://doi.org/10.1007/978-3-319-68152-8_5