THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

# Towards empirically grounded guidance for resource efficiency

## Applying, developing and synthesising environmental assessments

DANIEL BÖCKIN

Division of Environmental Systems Analysis
Department of Technology Management and Economics

CHALMERS UNIVERSITY OF TECHNOLOGY

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DANIEL BÖCKIN
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Division of Environmental Systems Analysis
Department of Technology Management and Economics
Chalmers University of Technology
SE-412 96 Gothenburg
Sweden
Telephone + 46 (0)31-772 1000

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# Abstract

Numerous solutions have been proposed to mitigate environmental damage, including resource efficiency and the vision of circular economy. Suggested solutions are often formulated as guidelines and heuristics like in the EU waste hierarchy, so-called R-hierarchies for resource efficiency and various guidelines for circular business models. However, these are often formulated on a conceptual basis without empirical support. Hence, it is often unclear in what contexts they are valid and how they can be interpreted for different types of products and applications. Systemic environmental assessments are necessary, and have been widely employed, to provide more solid empirical support for guidelines and for investigating the efficacy of suggested solutions. There is also a need for the results and learnings of those assessments to be easily understandable and usable for guiding decision-making towards reducing environmental impact within, say, product design and business management.

The purpose of this dissertation is to 1) formulate empirically grounded guidelines for resource efficiency and 2) test existing guidelines and heuristics in specific cases. The first aim is addressed by synthesising assessments of resource efficiency measures in literature. This revealed in what circumstances each measure can yield environmental benefits, depending on product characteristics, as well as when there are possible trade-offs and limitations. Several product characteristics were identified as of key importance for the efficacy of measures, including whether products are durable or consumable, active or passive, used for their full technical lifetime, used frequently or not and finally the product's complexity and pace of development.

The second aim is addressed by carrying out a prospective life cycle assessment (LCA) scrutinising the expectations of metal 3D printing for reducing automotive environmental impacts. The results showed that 3D printing can potentially reduce future life cycle impacts, by allowing redesign of components for lower weight and thus lower fuel consumption. However, this is only valid with low-fossil electricity for the printing process and developments towards printing with low-impact materials like low-alloy steel.

The second aim is further addressed by testing the potential environmental benefits of alternative business models. The method business model LCA method (BM-LCA) was developed for this purpose, taking the business itself as the object of analysis. The method uses economic performance as the basis of comparison, thus allowing a business to calculate the environmental consequences of business decisions. BM-LCA was applied to an apparel company, comparing selling and renting jackets. The results show that renting enabled sustained economic performance while reducing environmental impacts. This depended, however, on the sustainability of the transport and energy systems, as well as on business model parameters like price and rental efficiency, and on customer habits.

This dissertation shows that environmental assessments can be used to provide an empirical foundation for improved resource efficiency guidelines and to test the validity of heuristics Two key contributions and innovations are emphasised. The first is the formulation of empirically grounded guidelines based on key product characteristics. The second is the formulation and testing of BM-LCA, a method for assessing decoupling business from environmental impact.

**Keywords**: business models, circular economy, circular business models, decoupling, guidelines, life cycle assessment, resource efficiency, synthesis, waste hierarchy, 3D printing

# List of included papers

This dissertation is based on the work contained in the following papers:

I. Böckin, D., Willskytt, S., André, H., Ljunggren Söderman, M., Tillman, A.-M., 2020. How product characteristics can guide measures for resource efficiency: A synthesis of assessment studies. *Resources, Conservation and Recycling*, vol. 154C, pp. 104582. doi: 10.1016/j.resconrec.2019.104582

II. Böckin, D., Tillman, A.-M., 2019. Environmental assessment of additive manufacturing in the automotive industry. *Journal of Cleaner Production*, vol. 226, pp. 977-987. doi: 10.1016/j.jclepro.2019.04.086

III. Böckin, D., Goffetti, G., Baumann, H., Tillman, A.-M., Zobel, T., 2021. Business Model Life Cycle Assessment: A method for analysing the environmental performance of business (Manuscript submitted).

IV. Goffetti, G., Böckin, D., Baumann, H., Tillman, A.-M., Zobel, T., 2021. Using a novel LCA method for comparing business models: Learnings for business model innovation from a case study (Manuscript to be submitted).

# Contributions to included papers

I. AMT and MLS developed the idea and research design. DB, SW and HA performed the data collection and investigation, formal analysis of synthesizing data, documented metadata, and wrote the initial draft of the manuscript, in which HA wrote the introduction and aim, SW the method, and DB the analysis, results, discussion and conclusions. All authors developed the methodology and the analytical framework, critically reviewed and edited the manuscript. DB and AMT lead the extensive revisions of the manuscript to which SW, HA and MLS contributed with critical review and editing.

II. DB and AMT developed the idea and research design. DB performed the data collection, documented the metadata, developed the methodology, performed the formal analysis and wrote the initial draft of the manuscript, all supported by AMT. Both authors critically reviewed and edited the manuscript.

III. DB, GG, HB and AMT developed the idea and research design, supported by TZ. DB, GG and HB developed the methodology and performed the formal analysis. DB wrote the initial draft of the manuscript, supported by all other authors, who also critically reviewed and edited the manuscript.

IV. DB, GG, HB and AMT developed the idea and research design, supported by TZ, who also had initial contact with the company. DB and GG performed the data collection, documented the metadata and performed the formal analysis. GG wrote the initial draft of the manuscript, supported by all other authors, who also critically reviewed and edited the manuscript

# Other publications

Tillman, A.-M., Böckin, D., Willskytt, S., André, H. and Ljunggren Söderman, M., 2020. What Circular Economy Measures Fit What Kind of Product? Chapter in Handbook on the Circular Economy, M. Brandão, D. Lazaveric, G. Finnveden (eds), 2020, Cheltenham, United Kingdom: Edward Elgar Publishing Ltd.

Böckin, D., Goffetti, G., Baumann, H., Tillman, A.-M., Zobel, T., 2020. Environmental assessment of two business models - a life cycle comparison between a sales and a rental business model in the apparel sector in Sweden (2020:2). Gothenburg, Sweden: Chalmers University of Technology. https://research.chalmers.se/publication/519800

Tillman, A.-M., Ljunggren Söderman, M., André, H., Böckin, D. and Willskytt S., 2020. Circular economy and its impact on use of natural resources and the environment - Chapter from the upcoming book Resource-Efficient and Effective Solutions – A handbook on how to develop and provide them. Report no. 2020:1. Gothenburg, Sweden: Chalmers University of Technology.

Böckin, D., Willskytt, S., Tillman, A.-M. and Ljunggren Söderman, M., 2016. What makes solutions within the manufacturing industry resource efficient and effective? Paper presented at the 12th Biennial International Conference on EcoBalance. 3-6 October, 2016, Kyoto, Japan.

Willskytt, S., Böckin, D., André, H., Tillman, A.-M. and Ljunggren Söderman, M., 2016. Framework for analysing resource-efficient solutions. Paper presented at the 12th Biennial International Conference on EcoBalance. 3-6 October, 2016, Kyoto, Japan.

# Abbreviations

| | |
|---|---|
| BM-LCA | Business Model Life Cycle Assessment |
| CBM | Circular Business Model |
| $CO_2$ | Carbon dioxide |
| EC | European Commission |
| EMF | Ellen MacArthur Foundation |
| EU | European Union |
| IPCC | Intergovernmental Panel on Climate Change |
| LCA | Life Cycle Assessment |
| LCC | Life Cycle Costing |
| LCI | Life Cycle Inventory |
| MFA | Material Flow Analysis |
| PBF | Powder Bed Fusion |
| PCO | Product Chain Organisation |
| PSS | Product Service System |
| RE | Resource efficiency |
| RQ | Research Question |
| ResCoM | Resource Conservative Manufacturing |
| SBM | Sustainable Business Models |
| UNEP | United Nations Environment Programme |

# Preface

The illustration on the cover symbolises this dissertation's purpose. We all need guidance to lead us on the winding road up the mountain of resource efficiency and environmental sustainability. It is a peak that we must all reach together, and in this dissertation, I present my humble contribution to the effort.

From a personal perspective, the cover captures my essence in the depiction of a landscape that merges the dark green forests and lakes of my Swedish home with the clear blue skies and majestic towering mountains of my other homeland, Chile. The illustration also evokes a feeling of adventure. What is awaiting use on the peak? On the other side of the mountains?

The years since starting my PhD have indeed been an adventure, with twists and turns, setbacks and progress. The journey started by being accepted as a PhD student to the Mistra-REES research programme on *Resource Efficient and Effective Solutions based on circular economy thinking* (www.mistrarees.se). The programme is a collaboration between academia, large and small companies and societal actors, with the overarching aim of supporting the transition of the Swedish manufacturing industry towards a circular and sustainable economy. My role has been to investigate the efficacy of resource efficiency solutions in real world cases and formulate guidance towards environmental improvements. But at no point on my research journey have been alone, which brings me to the part where I want to thank everyone who have supported me on my journey, of which there are too many to bring up individually.

However, I have to make an exception for some. First and foremost, a massive thank you to Anne-Marie Tillman, who has supported me through thick and thin, who has challenged me, questioned me and praised me just when I needed it the most, and who has always been on my side and looked out for my well-being. You are a truly inspiring mentor!

Thank you also to Henrikke Baumann, my co-supervisor in the last part of my PhD, for thought-provoking feedback and brilliant insights and helping me cut through to the core ideas, which really helped in bringing out the best of me and my research.

I have collaborated with several other co-authors for this dissertation, to whom I am incredibly grateful. Thank you Hampus André and Siri Willskytt, for being excellent companions on this research journey. The frequently confusing PhD-life was easier to navigate together with you. Thank you also to Maria Ljunggren and Thomas Zobel, for invaluable inputs that elevated the papers, and by extension this dissertation, to reach as close as possible to its full potential. And thank you to Giulia Goffetti, for the countless sessions discussing the intricacies of Business Model LCA, the textile industry and of writing technical reports and scientific papers. A pandemic is a strange time to finish our PhDs, but I am glad that we could keep each other company!

I want to thank all of my colleagues from Environmental Systems Analysis, for creating an engaging and interesting work environment, I wish you all the best! Hanna Holmquist deserves a particular mention, for being a massive help in understanding the world of textiles and chemicals, and for lovely company, especially during lock-down times. I am immensely thankful to you for taking the time out of your hectic schedule of simultaneously parenting and dissertation-writing to help us when we needed it.

I also want to thank all of my colleagues at the Mistra-REES programme, for providing interesting perspectives, challenging questions and fun activities throughout the years, and particularly the programme director Mattias Lindahl. Additionally, a great thank you to those that have helped my research along the way, from sharing their expertise in interviews and study visits, to providing data and offering their valuable time to aid me in my research. Lisbeth Dahllöf deserves a particular mention, as do Valerie Boutty Gaillard and Damien Lemasson, without whom my first paper would never have come to be. Likewise, I am grateful to the informants at the (anonymous) case company for being open to our ideas and providing vital input and data that made my last two papers possible. And to my childhood friend, and talented artist, Petra Harmat Vergara, thank you for the lovely interpretation of my research that I am proud to put on the cover of my dissertation!

Most importantly, I want to say a deep thank you to my family, you have always been there for me, in the darkest times and in the good times, to offer well-needed support and guidance. Tack, y gracias!

And, finally, the biggest and most exciting twist in my PhD adventure came already on my first conference. That’s when I, by complete chance, ran into Dimitra Kopidou. Since then, every step of my journey has been with you beside me. You have believed in me, understood me, loved me, pushed me, comforted me and inspired me, for which I am eternally grateful. Σ’αγαπώ, ψυχή μου!

*Gothenburg, Sweden, March 2021*
Daniel Böckin

# Chapter 1 Introduction

Global resource use has increased dramatically in recent centuries along with the associated environmental impact, as documented in, e.g., UNEP (2017) and IPCC (2018). This development is tightly coupled with ever-increasing economic activity and consumption, as indicated by, for example, Blomsma and Brennan (2017) and Rockström et al. (2009). There are widespread efforts to mitigate such developments and various solutions have been suggested, both globally and for individuals, organisations and nations (IPCC, 2018; UNEP, 2017). This dissertation is concerned with one avenue for such action, namely improving resource efficiency, defined as minimising resource use in relation to a desired output (UNEP, 2010). ‘Resource use’ is interpreted as comprising both material resource consumption and environmental degradation (see section 2.1 for details). However, it is difficult to immediately discern the efficacy of solutions for mitigating environmental damage in real contexts (Blomsma and Brennan, 2017; Haupt and Zschokke, 2017; Kjaer et al., 2018). Several guiding visions and tools exist to aid design and decision-making. These include the vision of circular economy which aims to close material loops and maintain stocks of material using measures like reusing, remanufacturing and recycling (EC, 2020; EMF, 2013). Sometimes included within the umbrella of circular economy, and sometimes not, are measures for cleaner production and effective use (Ghisellini et al., 2016).

Measures for circular economy and resource efficiency are often systematised and formulated as guidelines or tools used to support decision-making for resource efficiency. These sometimes take the form of ranked lists of measures, of which a prominent example is the EU waste hierarchy (EC, 2008). Such guidelines go under many labels, including typologies, taxonomies, hierarchies, strategies, visions, tools or frameworks.[1]

Guidelines for resource efficiency have long been available – in 2003, for example, de Brito and Dekker presented something similar to what is now called ‘9R’ (see Cramer (2014) and Potting et al. (2017)) and the European waste hierarchy can be traced back to 1975 (EC, 1975). New guidelines are frequently proposed, and existing ones are often reformulated and updated. However, three problems with such guidelines can be identified. The first is that many of the available guidelines are formulated on a conceptual and top-down basis and require translation if they are to offer meaningful support to decision-making in specific contexts (Blomsma et al., 2019). Since the guidelines tend to be simple, and the suggested measures tend to rest on idealised descriptions of reality, it is often unclear in what contexts each guideline is valid and what limitations it may have (Ljunggren Söderman and André, 2019). One example of such a

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[1] Throughout this dissertation, the term ‘guidelines’ will be used, in order to emphasise the aspect of guiding decisions towards resource efficiency (for further clarification, see section 2.4).

guideline is the EU waste hierarchy, which explicitly calls for complementary assessments for more environmentally sound and factual decision support (EC, 2008).

The second problem relates to the lack of clarity in guidelines' intended application. It is often unclear how each guideline can be interpreted or operationalised in different areas of application (Blomsma, 2018; Blomsma et al., 2019). For example, the Ellen MacArthur Foundation's circular economy system diagram (or 'butterfly diagram') does not clearly indicate whether the proposed measures are useful for product designers, consumers, businesses or policy-makers (EMF, 2013). Furthermore, it is often unclear for what types of products the guidelines are valid. For instance, Yan and Feng (2013) do not specify for which types of products their 6R framework is valid, even though their suggested design for product modularity cannot be applied to, e.g., consumable products like food or fuel. Conversely, there are guidelines that are explicitly formulated for a specific sector or application but can be generalised to other sectors as well. For instance, Willskytt (2020) finds that design guidelines within one sector can be transferred to another sector if the products in question are handled by similar actors or have similar functions or legal requirements. For example, food and medicine have to be handled with similar hygiene requirements, which limits the possibility for reuse and recycling.

The third problem with many existing guidelines is associated with the ranking of measures that they recommend, such as the EU waste hierarchy that explicitly ranks, e.g., reuse higher than recycling (EC, 2008). A related example is the 'R-hierarchies' of different resource efficiency measures, in which it is more or less explicitly implied that 'smaller loops', such as reuse, are favoured over 'larger loops', like remanufacturing or recycling (Kirchherr et al., 2017; Reike et al., 2018). Such ranking can, however, be questioned in specific cases or contexts. For example, Ljunggren Söderman and André (2019) show that remanufacturing of electrical and electronic equipment offers limited benefits because it avoids material extraction but not the energy-intensive component manufacturing, why remanufacturing in this case is not always better than recycling. Cristóbal et al. (2018) provide an additional example, stating that the EU waste hierarchy does not provide enough guidance in the case of food waste prevention and management and that other tools are needed. Furthermore, measures are often interdependent, in which case ranking is not meaningful, as recognised by Blomsma and Brennan (2017) who introduce the concept of 'circular configurations' for combinations of measures working in sequence or in parallel.

Considering the problems associated with guidelines, there is a need to test them on real cases to achieve a more empirical basis for the expected gains in resource efficiency (Geissdoerfer et al., 2020; Manfredi et al., 2011; Rosa et al., 2019).

On a more specific level compared to systematised guidelines, there are informal heuristics, or rules of thumb, which are simpler statements or assumptions meant to guide environmental improvement in specific applications or contexts. For instance, it is often assumed that alternative business models like product-service systems (PSS) or circular business models (CBM) are inherently sustainable (Kravchenko et al., 2019). Yet in reality they may introduce trade-offs between different types of environmental impact and resource use, and may even increase resource use (Blüher et al., 2020; Chun and Lee, 2017; Mont, 2002). Another example is the heuristic in the automotive industry indicating that to reduce the environmental impact of vehicles, their weight should be reduced, which in turn reduces fuel consumption (Lewis et al., 2019). However, this could lead to trade-offs if lightweight components are less durable or require more environmentally damaging production (Lewis et al., 2019; Schau et al., 2012; Soo et al., 2016). Consequently, there is also a need for a more empirical basis for many heuristics, by testing the effects of their proposed solutions in real cases.

The solutions proposed by guidelines and heuristics can be achieved by various means. In particular, certain new technologies and new business models are often seen as important enablers for resource efficiency solutions. However, the actual benefits and drawbacks are uncertain, so there is a particular need for critical scrutiny. An example of such a new technology is digitalisation where, for example, Internet of Things is expected to enable sustainable solutions (Blüher et al., 2020; Ingemarsdotter et al., 2019) but may also increase energy use (Chen et al., 2020) and consumption of scarce resources that are challenging to recycle (Ljunggren Söderman and André, 2019). 3D printing is another new technology where there are considerable expectations of reducing component weight and hence fuel consumption (Lifset, 2017; Volvo Group, 2017). However, energy-intensive materials and printing processes can diminish the potential environmental gains (Gutowski et al., 2017). An example of a new business models CBMs, which are formulated as guidelines for businesses to operationalise the principles of the circular economy in order to decouple their economic performance from their environmental impact (Bocken et al., 2017; Geissdoerfer et al., 2017; Lüdeke-Freund et al., 2018). However, these guidelines are often formulated on a conceptual level that excludes crucial elements of the real business context, for example how a company will actually make money from the new business model (Blomsma et al., 2019; Pieroni et al., 2020). Similar examples include the business model strategies of Bocken et al. (2016) and the circular design framework of Moreno et al. (2016).

In summary, the validity of guidelines formulated on a top-down basis can be questioned, as can the validity of their ranking of measures for resource efficiency, and simpler heuristics in particular contexts. Additionally, it is often unclear how to operationalise guidelines in different areas of application. Systemic environmental assessments can be used to build more empirically grounded guidelines that avoid these problems, as well as to test the environmental efficacy of guidelines and heuristics in specific contexts (Manfredi et al., 2011). This dissertation will mainly utilise life cycle assessment (LCA), one of the most widely used methods for systemic environmental assessment. This method inventories all environmentally relevant flows related to a product or service are inventoried from cradle to grave, and then assesses them in terms of their effects on different categories of environmental impact (Baumann and Tillman, 2004; ISO 14040, 2006).

## 1.1. Purpose and research design

The purpose of this dissertation, in light of the research needs detailed above, is to achieve more empirically grounded guidelines and heuristics for resource efficiency. It will also test the environmental efficacy of heuristics in two cases with significant expectation of environmental improvement, namely 3D printing and alternative business models.

Based on this purpose, two research questions were formulated.

> RQ1: What guidelines for resource efficiency can be formulated based on existing assessment studies?

RQ1 was addressed by reviewing and synthesising numerous life cycle studies to produce a set of empirically grounded guidelines for resource efficiency based on product characteristics (Paper I). The studies were categorised according to a life cycle-based typology of resource efficiency measures, and synthesised to investigate the efficacy of each measure depending on product characteristics.

The second research question addresses the second part of the purpose, namely to test the environmental efficacy of existing guidelines and heuristics:

> RQ2: What environmental improvements can be achieved by realising guidelines and heuristics?

RQ2 can be addressed by carrying out quantified environmental assessments to test the efficacy of guidance given by heuristics and guidelines. In this dissertation, two particular cases were scrutinised. One concerns the emerging technology of metal 3D printing which is expected to revolutionise industry and contribute to environmental improvements, for instance in the automotive industry by reducing fuel and material consumption. This is addressed by using prospective LCA to assess the environmental benefits of applying metal 3D printing to a redesigned truck engine (Paper II).

The second case examined the expectation that new business models will enable decoupling of economic performance from environmental damage. This was investigated in two parts. First, a lack of appropriate methods for environmentally assessing and comparing business models was identified. Consequently, a method that builds on LCA but takes the business model itself as the object of analysis was developed (Paper III). Second, the method was applied to a particular case in order to investigate the environmental potential of a rental business model for jackets in the apparel industry (Paper IV). In practice, the investigations leading up to Papers III and IV were conducted simultaneously through an iterative process in which the case informed the methodological development and vice versa. The joint study with a full presentation of inventory and modelling data and a detailed application of the developed methodology was first published in a technical report (Böckin et al., 2020). The method was then refined and generalised in Paper III, while the assessment was tested and presented more clearly for a business audience in Paper IV.

Figure 1 provides an overview of how the overarching purpose of achieving empirically grounded guidance for resource efficiency was addressed. Papers II–IV are examples of case-based evaluations that can be used to assess the efficacy of guidelines or heuristics. Such assessments can also be synthesised in order to formulate new guidelines, as in Paper I.

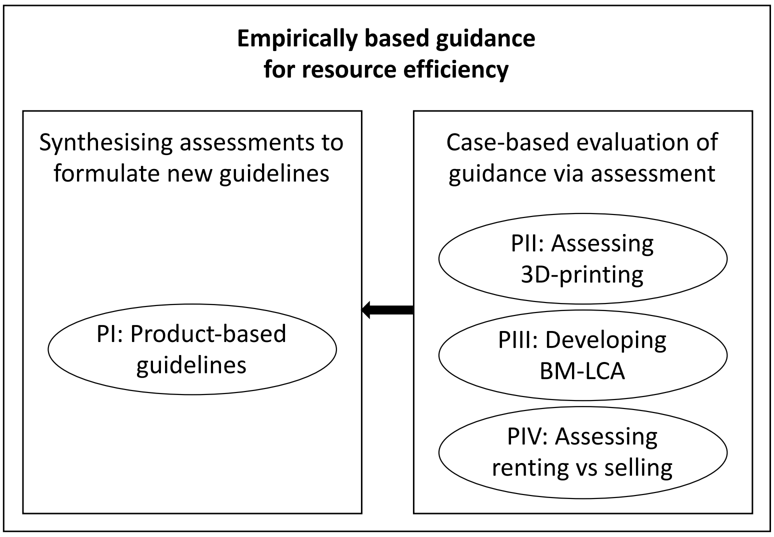

*Figure 1: Overview of how empirically-based guidance for resource efficiency can be achieved and how it is addressed in papers I–IV. The arrow indicates that case-based evaluations can be fed into syntheses, but note that, in this dissertation, only Paper II was among the studies synthesised in Paper I.*

The dissertation by outlining theories and concepts related to resource efficiency in order to present the current state of knowledge as regards the environmental efficacy of guidelines (Chapter 2). This is followed by a presentation of the methods used to carry out, develop and synthesise environmental assessments (Chapter 3). The research questions are addressed in three consecutive chapters: the synthesis of assessments (in Chapter 4), the prospective LCA case of 3D printing (in Chapter 5), and the presentation of BM-LCA and its application to a case of renting jackets (in Chapter 6). Chapter 7 discusses how the research in this dissertation contributes to knowledge on empirically grounded resource efficiency guidelines and the implementation of 3D printing technology and alternative business models. This is followed by a discussion on an actor perspective in environmental assessment and the validity and reliability of the dissertation’s findings. Lastly, Chapter 8 draws conclusions and outlines the scope for expanding and developing the research in this dissertation.

# Chapter 2 Theoretical and conceptual background

The following presents the background regarding resource efficiency and circular economy as responses to environmental degradation (section 2.1). This is followed by a conceptual background on business models and how they have might contribute to resource efficiency and circular economy (section 2.2). Then follow the theoretical foundations of tools for evaluating the consequences of solutions for resource efficiency, including industrial ecology and systems science (section 2.3). Finally, a literature background is presented on guidelines supporting decisions toward resource efficiency (section 2.4).

## 2.1. Resource efficiency and circular economy

The limits to growing global resource consumption have been expressed in early works like *The Economics of the Coming Spaceship Earth* by Boulding (1966) and *Waste Makers* by Packard (1960). Resource consumption has generated waste and caused environmental problems that have found increasing awareness after such works as *Silent Spring* by Carson (1962) and *Limits to Growth* by Meadows et al. (1972). Today, there are many responses and proposed solutions to these environmental and resource problems.

One type of response can be termed '*resource efficiency*'. Throughout this dissertation, the term 'resource' is used as a synonym for 'natural resources', defined as assets that occur in nature, from which they are extracted to be used for human purposes in society (Tillman et al., 2020). In this instrumental view, such assets include renewable and non-renewable resources as well as ecosystem services provided by the natural system (including provisioning services, regulating services, cultural services, and underlying supporting services). Consequently, 'resource use' refers to both material resource consumption and environmental degradation. 'Resource efficiency' thus means minimising the use of resources in relation to a specific desired output, or in other words, 'resource efficiency enhances the means to meet human needs while respecting the ecological capacity of the earth' (UNEP, 2010).

The concept of circular economy is a subset of resource efficiency and is an umbrella concept for solutions that recirculate and extend the use of products, components and materials (Blomsma and Brennan, 2017; Kirchherr et al., 2017). The circular economy has been high up on the political agenda in recent years, for example in Europe (EC, 2015, 2020) and in China (Yong, 2007; Zhijun and Nailing, 2007). Furthermore, the concept has garnered much attention and expectation within industry (EMF, 2015b; Ghisellini et al., 2016).

Ultimately, the circular economy aims to achieve resource efficiency by altering the physical flows of material and energy throughout society and nature. To achieve this, concrete physical measures must be implemented. There is a plethora of such physical measures in literature, for circularity and for resource efficiency in general, with varying and overlapping definitions.

One way to organise these physical measures, presented in Paper I, is according to the part of the product life cycle that they address, be it extraction and production, use or post-use (commonly referred to as end-of-life). The typology presented in Figure 2 offers a comprehensive overview of the possible measures that can be adopted for resource efficiency in different life cycle stages. The purpose of the typology is mainly to act as an organising structure for measures (see section 3.3), as opposed to acting as a guideline on its own. While the typology presented was designed by the authors of Paper I, and is organised according to a product life cycle, it draws on existing formulations of measures in the circular economy literature (Allwood et al., 2011; EC, 2008; EMF, 2013; Potting et al., 2017; Stahel, 2010; Stahel and Clift, 2016). This was complemented by definitions of remanufacturing (Sundin, 2004) and functional recycling (Graedel et al., 2011; Guinée et al., 1999). Furthermore, the typology draws on eco-design literature such as the ten golden rules (Luttropp and Brohammer, 2014), the eco-design strategy wheel (Brezet and van Hemel, 1997) and other eco-design guidelines as described by Ceschin and Gaziulusoy (2016) and Sundin (2009) for example.

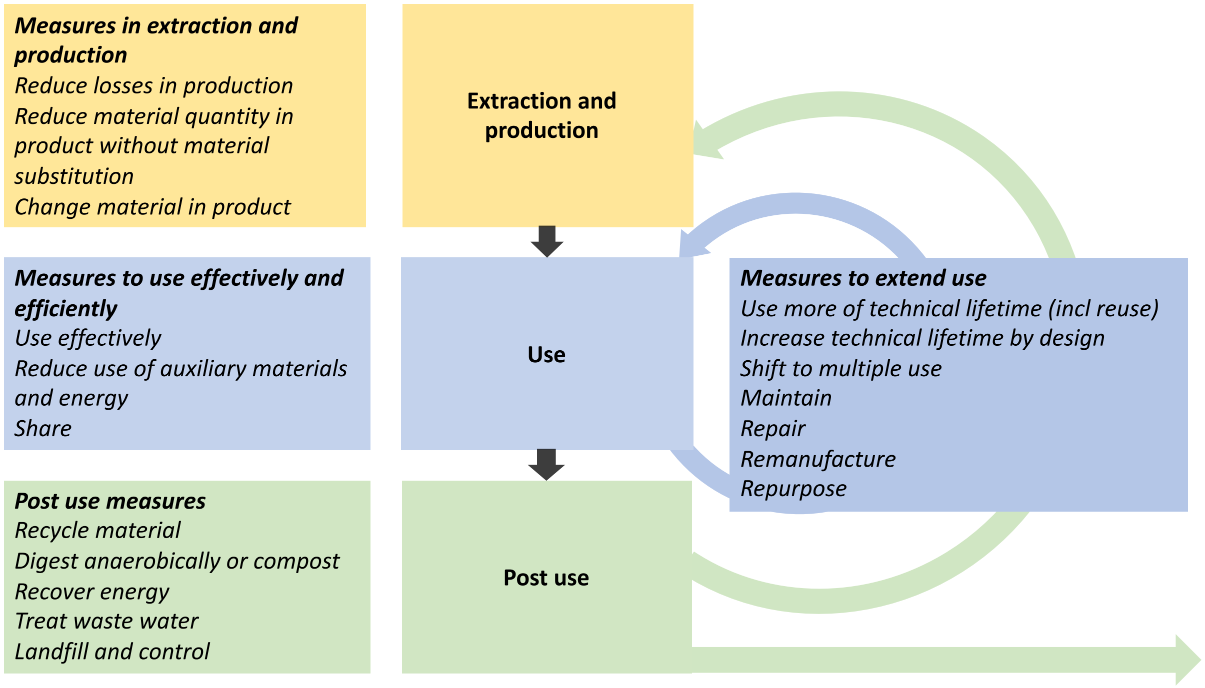

*Figure 2: Typology of physical measures for resource efficiency, from Paper I.*

Concrete descriptions of how circular economy and resource efficiency can be physically achieved are given here, based on the sources mentioned above. Starting from the life cycle stages of **extraction of raw materials and production of materials and products**, resource efficiency can be accomplished by *reducing losses* of material or energy in production, for example by reintroducing scrap and energy flows into the production process or by valorising them in other production chains, through industrial symbiosis or process integration. The *quantity of material can be reduced*, while still using the same material in the product.

The *material composition of products can be changed*. For example, fossil, hazardous or scarce materials can be substituted, and recycled material can be used instead of primary material. Material substitution can increase resource efficiency in itself (e.g., through excluding hazardous constituents) or enable other measures (e.g. increase technical lifetime through increased durability).

The use of a product can be improved in two principal ways, through **using the product effectively** and efficiently and through **extending its use**.

*Use effectively* means to deliver (which is relevant to the provider) or acquire (which is relevant to the customer) function according to the user's needs but not more. An example is the smart dispensing of soap. Effective use also includes making sure the product is used for its intended purpose. Effective use might increase the functionality of products in order to improve system efficiency, such as detergents allowing for lower washing temperatures. *Reduced use of auxiliary materials and energy*, such as energy efficiency improvements, also belongs to the group of use-phase efficiency, as does *sharing* a product between several users. Using a dissipative product effectively is analogous to efficient use of the corresponding active product, e.g., when it comes to water use (dissipative product) in a building (active product).

To **extend the use** of products means to prolong their lifetime. This can be done by *using more of the technical lifespan* of the product, by the same user or a new one (the latter is often denoted reuse). The product may also be redesigned for *increased technical lifetime*, and a disposable product can be redesigned to become a *multiple-use product*. The use of a product may also be extended through restorative interventions such as *maintenance*, *repair*, *remanufacturing* or *repurposing*. *Maintenance* involves activities where products are inspected, maintained and protected before breakdown or other problems occur. *Repair* takes place after wear, malfunction or failure. *Remanufacturing* is the process of restoring a product to a state as good as new or even better, through disassembly, repair or exchange of components, re-assembly and quality assurance. *Repurposing* means reuse of a product in a different function than the original function.

The last category, **post-use**, addresses the end-of-life of products and components. *Recycling* recovers and returns materials to use. In recycling without quality loss, the properties and function of a material are maintained. Thus, the recycled material can replace virgin raw materials and be used for the same function. However, recycling usually leads to quality loss, in which the material properties (and hence also function) deteriorate.

Biodegradable materials can be *digested anaerobically* or *composted* (yielding, e.g., biogas, recovered plant nutrients and landscaping material). *Energy recovery* converts the energy stored in materials into usable energy carriers such as heat and electricity. *Landfills* are constructed to limiting the environmental impact of disposing of discarded products and may include landfill gas collection for energy recovery.

Note that design changes are a necessary precondition for many, if not most, of the measures in the typology. Consequently, design is not included as an explicit measure in the typology but is instead an inherent aspect of most measures. Furthermore, each physical measure can be achieved in different ways, such as by adopting new business models, which are further addressed in this dissertation, so a corresponding background follows here.

## 2.2. Business models and sustainability

The business model concept has many interpretations, often diverging (De Angelis, 2018; Zott et al., 2011). The Cambridge Dictionary defines it as 'a description of the different parts of a business or organization showing how they will work together successfully to make money' (www.dictionary.cambridge.org). A definition commonly used in literature is that 'a business model describes the rationale of how an organization creates, delivers, and captures value' (Osterwalder and Pigneur, 2010). In this description, an organisation uses various resources and activities to create value based on an offer to its customers. Then it will use its available channels to deliver that value to its customers. Value capture is the process of making money from that value delivery. Taken together, the business model can be seen as the scheme according to which a business strategy will be implemented throughout an organisation (Osterwalder and Pigneur, 2010).

Historically, the business model concept was popularised in the 1990s as a way to summarise or simplify the new ways of creating, delivering and capturing value that were enabled by a booming e-commerce sector (Geissdoerfer et al., 2018a; Magretta, 2002). Subsequently, business models have been used by scholars and business strategists to explain a company's performance and competitive advantage. They have also been used by companies, either as a subject of innovation in itself, or as a way to commercialise new technologies (Zott et al., 2011). The business model concept has also been applied to investigate how companies can contribute to environmental sustainability (Massa and Tucci, 2014; Nußholz, 2020).

In a so-called linear business model, a company captures value (i.e. generates profit) via the continuous sale of products. Alternative business models have been formulated to potentially reduce the environmental impacts of businesses, such as sustainable business models (SBM), product-service systems (PSS) and circular business models (CBM). In brief, SBMs are defined by Geissdoerfer et al. (2018b) as 'business models that incorporate pro-active multi-stakeholder management, the creation of monetary and non-monetary value for a broad range of stakeholders, and hold a long-term perspective'. PSS is a subset of this and is defined by

Tukker and Tischner (2006) as ‘a mix of tangible products and intangible services, designed and combined so that they are jointly capable of fulfilling final customer needs’. Finally, CBMs are another subset of SBMs, that partially overlaps with PSS, which lack a universally agreed upon definition (Geissdoerfer et al., 2018b) but are defined by Linder and Williander (2017) as ‘a business model in which the conceptual logic for value creation is based on utilizing economic value retained in products after use in the production of new offerings’.

A variety of studies have been carried out relating to the environmental assessment of alternative business models, although most actually compare product alternatives that represent different business scenarios. Examples include environmental LCAs of renting next-to-skin garments (Bech et al., 2019), water purifiers (Chun and Lee, 2017), strollers (Kerdlap et al., 2021) and power-tools (Martin et al., 2021) as well as of clothing libraries (Zamani et al., 2017). The same is true in the case of LCAs that are complemented by qualitative business considerations, such as assessments of cloth diapers (Hoffmann et al., 2020), energy storage technologies (Tschiggerl et al., 2018) and veterinary pharmaceutical products (Barbieri and Santos, 2020).

A limited number of studies have attempted to add economic considerations to environmental assessments by modelling. The work of Asif et al. (2016) is an example attempting integrated modelling of environmental and economic effects. They developed a tool based on system-dynamics and agent-based modelling to assess leasing of washing machines, although the basis of comparison is the product function rather than the business itself. Further studies that base their assessment on product comparisons include those that apply LCA and life cycle costing (LCC) in parallel. Examples include studies comparing plug-in and wireless charging for electric buses (Bi et al., 2017; Bi et al., 2015) and studies on PSS models for passive durable products like furniture and exhibition equipment (Kaddoura et al., 2019), energy-using equipment for separating air into its constituents (Zhang et al., 2018) and eco-efficiency calculations for disposable diapers (Mendoza et al., 2019).

### 2.3. Industrial ecology and systems science

Holistic assessments of environmental consequences are central to this dissertation, both in the implementation of physical resource efficiency measures and alternative business models. The field of *industrial ecology* provides tools for such evaluations (Graedel and Allenby, 2010). Industrial ecology is an interdisciplinary field based on the analogy of technical systems with ecosystems, where industrial activities are assessed in relation to the natural system’s capabilities, with the purpose of identifying solutions for long-term sustainability (Andrews, 2000; van Berkel et al., 1997). Tools like LCA, LCC and material flow analysis (MFA) can all trace their origins to industrial ecology and are thus methods that take a systems perspective in assessing environmental, economic and material flows.

The systems perspective is in turn rooted in *systems science*, an inherently interdisciplinary science with roots in disciplines ranging from thermodynamics, biology and ecology to cybernetics and control engineering (Ingelstam, 2002). A ‘system’ in this context can be defined as a number of components within a system boundary, connected through some interrelations (Boulding, 1966; Churchman, 1967). Studying a system means understanding it as more than simply a sum of its parts; the connections between components are especially relevant (Churchman, 1967; von Bertalanffy, 1969). In other words, where more traditional natural and social science is reductionist and mechanistic and focussed on analysis, systems science is instead holistic, with an emphasis on synthesis. Rather than seeking deep understanding of the details, systems science pursues the development of system models, typically with the purpose of not only seeking understanding of components and interrelations but of solving a real-world problem or improving a situation (von Bertalanffy, 1969). In the efforts to solve such problems, inspiration is taken from a wide range of disciplines. Any available tool that suits the purpose is used to study the system at hand and arrive at an answer to the questions posed. Throughout this dissertation, the real-world issues considered are the environmental challenges faced globally as well as the guidance of different actors in the choice and implementation of resource efficiency solutions.

## 2.4. Guidelines for resource efficiency

As mentioned in Chapter 1, decision support for resource efficiency takes on many forms. Before reviewing existing literature on the topic, however, it is relevant to clarify the terminology around such guidance, including, for example, typologies, taxonomies, strategies, hierarchies, tools, measures, visions and frameworks, which are often used inconsistently (Schöggl et al., 2020). Inspired by the review of Blomsma (2018), guidance can be roughly divided into three levels of detail. While each level may go under various names, the terms chosen for this dissertation will be defined here. The highest level is termed ‘visions’, defined as overarching concepts like sustainable development (Brundtland, 1987) or circular economy (EMF, 2013). The second level can be termed ‘frameworks’, defined as more detailed proposals of how such visions should operate on an overall level and ‘how it can support high level courses of action’ (Blomsma, 2018). Examples include the performance economy of Stahel and Clift (2016) and the circular economy system diagram (or ‘butterfly diagram’) of EMF (2013). A subset of the second level is ‘guidelines’, here defined as operationalisations of frameworks, that are intended to act as decision support for various applications like policy-making or product design. Guidelines for resource efficiency often take the form of ranked sets of solutions or measures, like the EU waste hierarchy (EC, 2008) or the ReSOLVE guidelines (short for regenerate, share, optimise, loop, virtualise and exchange) by the Ellen MacArthur Foundation (EMF, 2015a). The third level is termed ‘measures’, which are the individual solutions proposed by the guidelines, such as prevention in the EU waste hierarchy or sharing in ReSOLVE. Note that the typology of physical measures presented in section 2.1 is not meant as guidance and does not fall into any of these three categories.

This dissertation is mainly concerned with the level of guidelines, as well as of specific measures for resource efficiency. Individual measures can be tested using environmental assessments in individual cases, introduced in section 3.1. Here will be presented a background on the guidelines that have generalised and systematised measures for resource efficiency.

Guidelines can roughly be divided into two categories. First are general guidelines where the intended application is unspecified or extremely wide. These include the typology of solutions proposed by Reike et al. (2018). They review literature on 'R-hierarchies' (a collective name for 3R, 6R, 9R etc.) which informs the creation of their typology of 10R with clear rankings between each measure. Additional examples of general guidelines include the waste hierarchy (EC, 2008) and ReSOLVE (EMF, 2015a), also mentioned above.

The second, and more common, category includes guidelines that apply to more or less specific intended applications. In some examples, the intended application is stated in vague terms, as with the concept of resource conservative manufacturing (ResCoM) by Rashid et al. (2013) which is formulated to achieve resource efficient conceptual design in manufacturing, the 9R strategies by Potting et al. (2017), intended to guide policy-making.

Other examples of guidelines for design applications include the ReX taxonomy by Sihvonen and Ritola (2015), which is formulated in support to the product development process. Similarly, the 6R-hierarchy by Yan and Feng (2013) is intended to support product design towards resource efficiency, while the 3R hierarchy of Gehin et al. (2008) is meant for the early stages of product design. Further, as a contrast to R-hierarchies, Vezzoli and Manzini (2008) formulate design guidelines that avoid a general ranking of measures by proposing different guidelines for products with different characteristics. Likewise, Willskytt and Brambila-Macias (2020) build on Paper I in this dissertation to formulate eco-design guidelines for resource efficient products depending on product characteristics. A large variety of guidelines specific to eco-design exists, as reviewed by Pigosso et al. (2015) and Rossi et al. (2016), but will not be detailed further here, since product design is not a central focus of this dissertation.

There are several guidelines that aim to address business model innovation. Building on previous work by Bocken et al. (2014) and Bakker et al. (2014), Bocken et al. (2016) present a set of CBM strategies to 'give clarity and direction to designers and strategic decision makers in businesses that want to pursue a CBM'. Inspired by Stahel (2010), they introduce a taxonomy of *slowing*, *closing* and *narrowing* resource loops.[2] Slowing loops means prolonging the use and reuse of products over time. Closing loops means reusing materials by recycling. Narrowing loops means reducing the resource use related to the product and production process. Based on the authors' taxonomy, they formulate a set of CBM strategies to act as conceptual aids in product design and business model innovation. These strategies relate to slowing and closing loops, while narrowing loops is excluded since it does not relate directly to the cycling of resources. For slowing loops, their suggested business model strategies include the access and performance model (where users share products or pay per function), extending product value (prolonging the period when products can provide their function), the classic long-life model, and a model based on encouraging sufficiency. Strategies for closing loops include extending resource value and industrial symbiosis.

Further examples of guidelines are the CBM patterns developed by Lüdeke-Freund et al. (2018), who review literature to formulate a tool supporting the implementation of CBMs. From their analysis they identify six generic CBM patterns, namely: 1) repair and maintenance, 2) reuse and redistribution, 3) refurbishment and remanufacturing, 4) recycling, 5) cascading or repurposing and finally 6) organic feedstock. Additionally, they link these patterns to design strategies and indicate the type of resource savings that can be achieved in each case, although without detailing how or to what extent this happens.

While the CBM guidelines mentioned so far tend to be lists of physical measures, there are guidelines that take business considerations into account more explicitly. An example is product-service systems (PSS), defined as 'a mix of tangible products and intangible services, designed and combined so that they are jointly capable of fulfilling final customer needs' (Tukker and Tischner, 2006). PSS solutions are categorised as product oriented (based on the sale of products with additional services), use oriented (where the provider keeps ownership of the product) or result oriented (where a function or result is provided without specifying which products may be involved). Result-oriented PSS are expected to have the highest potential to reduce environmental impacts (Tukker, 2015).

Pieroni et al. (2020) identify and consolidate business model archetypes into a typology of CBMs to aid in circular business model innovation. To this end, they follow the terminology of Urbinati et al. (2017) who distinguish business models based on where and how resource decoupling is achieved. Business models that alter value creation, for example via product design or reverse logistics, are categorised as *upstream*. These capture value mainly through reductions in costs or in raw material input or waste output. Upstream business models include those that address circular production and distribution, as well as circular sourcing. On the

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[2] Note that the work in this dissertation does not follow this terminology, but rather follows the definition of resource efficiency given in section 2.1.

other hand, *downstream* models are those that alter value capture or delivery by introducing new revenue schemes or customer interfaces. Value capture occurs mainly through additional revenues, market penetration or brand enhancement, as well as from increased utility, and longevity of products and materials. Downstream models are those that address dematerialisation, collaborative consumption, PSS and long life. The guidelines are formulated as an aid to qualitative ideation and innovation of business models towards circularity, rather than for evaluating which models are preferable from a resource efficiency perspective.

Whalen (2019) formulates recommendations for how firms with business models based on extending product value can contribute to resource efficiency. Based on the level of interaction between the firm and the product in question, she distinguishes three archetypes. 'Facilitators' provide a platform for their customers to exchange products that the firm itself has no interaction with. 'Redistributors' instead collect, repackage and sell products that would otherwise go to waste. In contrast to the first two, 'doers' carry out the life-extending activities on their own products. It is suggested that doers and redistributors can contribute more to resource efficiency than facilitators, where replacement rates are expected to be lower.

More specific to manufacturing companies, Blomsma et al. (2019) create a circular strategies scanner that aims to support the early phases of circular innovation by enabling the translation of circular economy concepts into practice. Their tool allows the systematic exploration of circular strategies, in order to map a manufacturer's existing strategies or to generate ideas for new ones. Furthermore, their tool connects strategies with each other to help identify synergies and trade-offs.

In the case of consumer goods, Moreno et al. (2016) formulate design guidelines based on CBM archetypes that attempt to connect circular design with circular business model innovation. Their CBM archetypes include those that address 1) circular supplies, 2) resource value, 3) product life extension, 4) extending product value and 5) sharing platforms. Considering the type of value creation that each archetype offers, they then propose circular design strategies suitable for each business model.

Guidance for more specific sectors include, for example, studies that use literature and stakeholder input to develop green principles for lightweighting of vehicles (Lewis et al., 2019), for battery management in grid applications (Arbabzadeh et al., 2016) and in electric vehicles (Arbabzadeh et al., 2019).

# Chapter 3 Methods

The research presented in this dissertation was carried out using several different methodologies. Central to each of these approaches are empirical case studies in the form of environmental assessments. A case study examines phenomena in their real-world contexts (Siggelkow, 2007; Yin, 1981), and achieves generalisation, not by statistical analysis, but through in-depth analysis and understanding of the studied phenomena (Flyvbjerg, 2006; Gibbert et al., 2008). Although single case studies have limited use in formulating general theory or general recommendations, they can instead highlight details and connections in the specific case that are not expressed or captured by the general theory (Siggelkow, 2007). Furthermore, a single case can falsify (or support the validity of) the general theory in that particular context (Flyvbjerg, 2006). An example in the context of this thesis is that a case-based environmental assessment can contradict general guidelines for resource efficiency in a specific case.

While individual assessment studies provide in-depth answers and decision support for the questions raised, there is also a complementing way to achieve more breadth. Synthesising the results and learnings from existing studies can provide this breadth (Flyvbjerg, 2006), while ensuring that any decision support is empirically grounded.

Further uses for empirical case studies include acting as a support or foundation for methodological development (Tillman, 2012). The results of several cases can be synthesised to achieve more generic knowledge. Figure 3 illustrates the interplay between empirical cases, method development and synthesis. Starting in the upper left of the figure, an empirical case is carried out using some research approach in order to answer a question from practice. For example, a study using LCA may set out to determine whether option A or B is environmentally preferable. The immediate outcome of the assessment shows which option is preferable. However, in carrying out the study, limitations of the research approach are often identified which may lead to development of new methods or adaptations of existing methods. The new method may in turn be applied in a new empirical case, answering a new question from practice. The pattern may potentially repeat indefinitely. The bottom of the figure shows a separate or related research stream that follows a similar pattern.

In summary, all methods employed in this dissertation relate to environmental assessment, more specifically carrying out individual assessments in the form of LCA (introduced in section 3.1), developing a new assessment methodology based on an empirical case (introduced in section 3.2) and synthesising existing assessments (introduced in section 3.3).

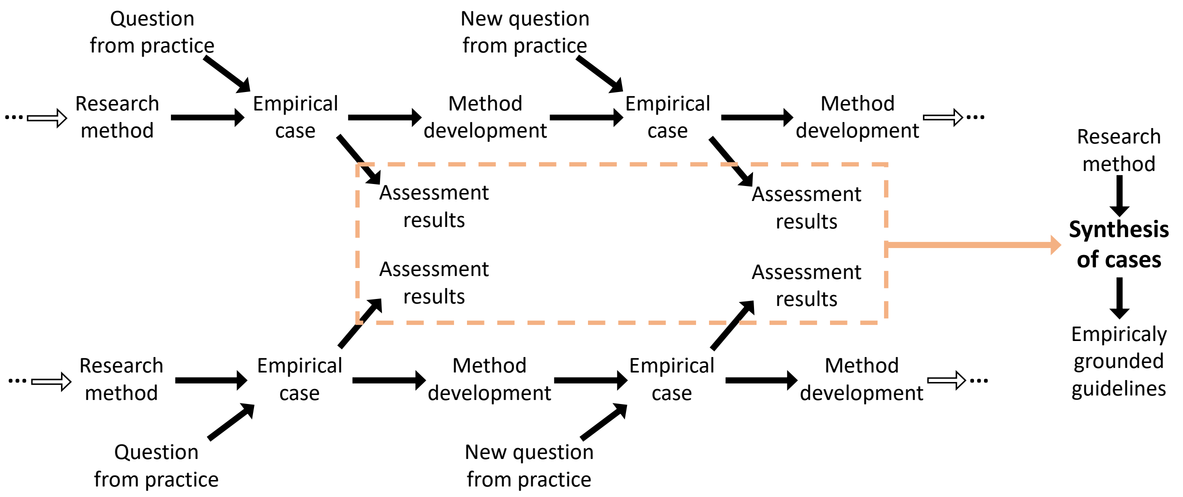


*Figure 3: Illustration of the interplay between empirical cases, method development, assessment results and the synthesis of these results into guidelines providing empirically grounded decision support. Adapted from Tillman (2012).*

### 3.1. Quantitative environmental assessments

The environmental and resource implications of resource efficiency measures can be quantitatively assessed by investigating and comparing the resulting changes in physical flows of material and energy. Keeping a systems perspective gives a holistic view of the system studied, which can, for example, reveal burden shifting between life cycle stages and environmental impacts. Tools for this include MFA (Ayres and Ayres, 2002) and LCA, which is the principal assessment method with which this dissertation is concerned. LCA is a well-established methodology for quantifying all relevant inputs and outputs of material and energy related to a product system in order to evaluate environmental impacts. The results are interpreted in relation to the objectives of the study (Baumann and Tillman, 2004; ISO 14040, 2006).

There are four main phases in LCA that are typically carried out iteratively, namely 1) goal and scope definition, 2) life cycle inventory analysis, 3) life cycle impact assessment and 4) interpretation. In the first phase, the purpose of the study is stated, and environmental impact categories and system boundaries are chosen. If relevant, a choice is made of which alternatives to compare. Most importantly, a functional unit must be defined, which is the reference unit to which all flows are related, and which serves as the basis of comparison between alternatives. The functional unit is commonly defined as a physical characteristic describing the function of the product in question, for example, ‘1 litre’ for packaging, ‘1 tonne*km’ for goods transportation and ‘1 m2*year’ for surface materials like paint or flooring.

The second phase, life cycle inventory (LCI) analysis, entails constructing a system model which is an incomplete mass and energy balance that only considers environmentally relevant flows. The system model describes the processes in the life cycle and the flows between them. In guiding the choice of the type of data to use, it is useful to divide processes into foreground and background processes (Tillman, 2000). Foreground processes are those that are directly affected by decisions on the system under scrutiny. Hence, data for foreground processes should preferably be collected from primary and site-specific sources (Earles and Halog, 2011). In contrast, background processes, such as the energy or transportation system, are only indirectly affected by decisions on the analysed system. As such, it is often enough to collect average industry data for these processes. It is worth noting that the main effect of an analysed measure is often found in the background system, so it is often crucial to choose appropriate and relevant background data. In general, the data can be collected from various sources such as directly from the industries/actors that are being analysed, for example, from expert interviews or measurements. In many cases, such data are already available in databases or presented in literature. Once the appropriate data have been collected, the quantity of each flow is scaled according to the functional unit.

Subsequently, in the third phase of life cycle impact assessment (LCIA), the flows quantified in the second phase are aggregated according to how they affect different categories of environmental impact. Generally, this is done by applying ready-made LCIA methods, which are natural science-based models of cause-effect chains, quantifying how important different emissions are for specific categories of environmental impact. Potential environmental effects from emissions that contribute to a particular type of impact are thus quantified and aggregated into the corresponding category. The potential impact in this category is then expressed as a score for an indicator, such as global warming potential (in CO2-equivalents) for climate change or dissipated water (in m3) for water consumption. Scores expressed with such indicators are at the ‘midpoint’ level. The cause-effect chains of emissions and resource consumption can be further followed to their so-called ‘endpoints’, which reflect the damage on three areas of protection, namely human health, ecosystem quality and natural resources. Endpoint indicators are thus more relevant to the subjects that are deemed worthy of protection. However, such modelling also brings inherent uncertainties because of the more complex modelling of longer cause-effect chains.

All impacts can be aggregated into a single score by weighting the different types of impact according to their perceived relevance (Pizzol et al., 2017). While such weighting necessarily includes value judgements and entails the loss of nuance and detail in the results, it can give a useful overview for studying relative results between different options. Different weighting methods are based on different values, assumptions and logic (Hauschild and Potting, 2005). Consequently, each method tends to emphasise different aspects of the LCI. As such, weighting can also be used to filter the results from the LCI and identify key indicators to be further analysed and presented in depth. Such a procedure was first described by Tillman et al. (1998), who used several distinct weighting methods in a first step to filter the results and thus identify the impacts or emissions that dominate the results in one or more of the employed weighting methods. This approach is used in the assessments in Papers II and IV.

Finally, the fourth phase of LCA is interpretation, which is the process of reaching conclusions and recommendations by analysing the results and their robustness, and considering the pros and cons of any compared alternatives. Extensive interpretation is carried out in Papers II and IV.

When applying LCA to particular systems, like emerging technologies or business models, there are specific considerations to be aware of, which will be presented in the following sections.

### 3.1.1. Prospective LCA

While the original aim of LCA was the environmental assessment of existing systems from a short-term time perspective, the potential environmental consequences of emerging technologies should also be investigated (Sandén, 2004). The purpose of a prospective LCA is to assess the potentials and risks of such emerging technologies (Arvidsson et al., 2017). Prospective LCAs are inherently uncertain but can be used to guide technological development in a desired direction, for instance to minimise environmental impacts (Villares et al., 2017). Technological development can have effects directly on the foreground processes being investigated, like improved energy efficiency for the product, but also on background processes like electricity or transportation systems which can change over time (Arvidsson et al., 2017).

In Paper II, a prospective approach to LCA was employed in the case of 3D printing in the automotive industry. Potential future effects were taken into account by formulating two scenarios, for the present and future states of metal 3D printing, respectively, in addition to a reference scenario (see Chapter 5). The future scenario was placed roughly a decade in the future. Thus, several potential technical performance factors had to be estimated, such as the possible size of components that can be 3D printed and what materials will be available for printing. Furthermore, a future low-fossil electricity supply was assumed. In addition to using scenarios, the uncertainties of prospective LCA were addressed by making conservative assumptions and carrying out sensitivity analyses to improve the reliability of the results.

### 3.1.2. LCA of business models

LCA applied to business models requires additional considerations compared to more mainstream LCA. The key difference is that the object of analysis in mainstream LCA is typically a product or service, which means that the studied system is purely technical. When instead a business model itself is taken as the object of analysis, this requires that more than material and energy flows are taken into account. Socio-technical factors need to be considered, like the interactions of a business' with its value chain and economic dimensions, need to be considered. These are seldom included in LCA (Costa et al., 2019), why there is no established method for doing so. As expressed in Paper III, if such an approach were to be useful in guiding business decisions towards reducing the environmental impact associated with its economic activities, these business decisions should be coupled to their potential environmental consequences by modelling the physical and monetary flows.

To achieve such modelling, it is necessary to understand what parts of the product system belong to the company operating the business model, and their relation to the surrounding network of actors, including, including suppliers and customers. An approach based on product chain organisation (PCO) can be adopted to do this (Baumann, 2008, 2012; Lindkvist and Baumann, 2017). So-called 'socio-material interaction points' between actors define the material and environmental flows to be assessed. A transaction between two actors implies the exchange of money and goods and/or services as seen in the simple schematic in Figure 4. It is worth noting that, in the case of services, there is almost always some associated use of materials and energy. An environmentally and economically integrated assessment then has to investigate how these transactions influence a business model's environmental impact. Furthermore, in order to achieve a fair comparison of business models, their function must be identified. While standard LCA uses physical product characteristics to express the function (see section 3.1), a business model's function should, in order to be useful for businesses, be expressed in a way that represents economic performance. These methodological considerations were implemented in a developed method, presented in Paper III, called the business model LCA (see section 6.1).

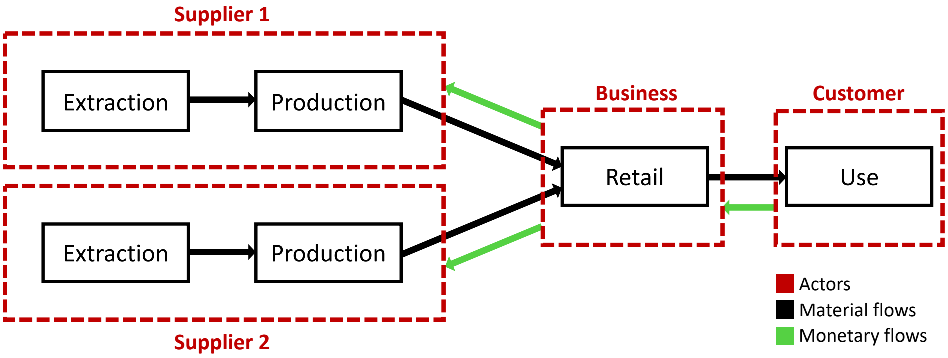


*Figure 4: Simplified flowchart showing the transactions between actors in a value chain, with the associated material flows in one direction and monetary flows in the other (Paper III).*

### 3.2. Developing new assessment methodology

In addition to answering specific questions from practice (see Figure 3), empirical case studies can also reveal shortcomings in existing methods, thus prompting method development. Paper III identified the lack of assessment methods that take business models as the object of analysis. Following the principles for LCA of business models described in section 3.1.2, such a method was developed and presented in Paper III and applied to a case in Paper IV. The method development followed an iterative process. This entailed alternating between applying the principles, working out the monetary and material flow relationships of the business models and further developing the method based on insights from the case studied. The notion to shift from a product focus to a business focus was realised before the case study. The case was then used to obtain some real socio-material interactions and to model and quantify these, in order to concretise the coupling between physical flows and money flows and define a profit-based functional unit. The case study was a comparison of two business models for a Swedish apparel company, a sales model for jackets and a rental model for the same jackets. This was presented first in a technical report (Böckin et al., 2020), and then framed in the context of business model innovation in Paper IV). Learnings from the case informed the method formulation which in turn informed the environmental assessment in the case.

Once the method was functional and consistent, it was generalised to enable its application on other business models and presented in Paper III, which is summarised in section 6.1.

## 3.3. Synthesis of assessments of resource efficiency measures

While guidelines can be formulated as ranked lists of measures, the point of departure for the synthesis in Paper I was the hypothesis that product characteristics are a more relevant foundation for formulating recommendations for resource efficiency. This hypothesis was based on co-authors' many years of experience within the field of LCA as well as insights from other attempts to formulate guidelines in the literature (Vezzoli and Manzini, 2008). In order to formulate product-based guidelines, the results of assessments of resource efficiency (RE) measures were systematically synthesised, following a procedure schematically presented in Figure 5. The first step is to collect relevant assessment studies from literature, then analyse and categorise each study according to an analytical framework, and lastly to synthesise all collected studies.

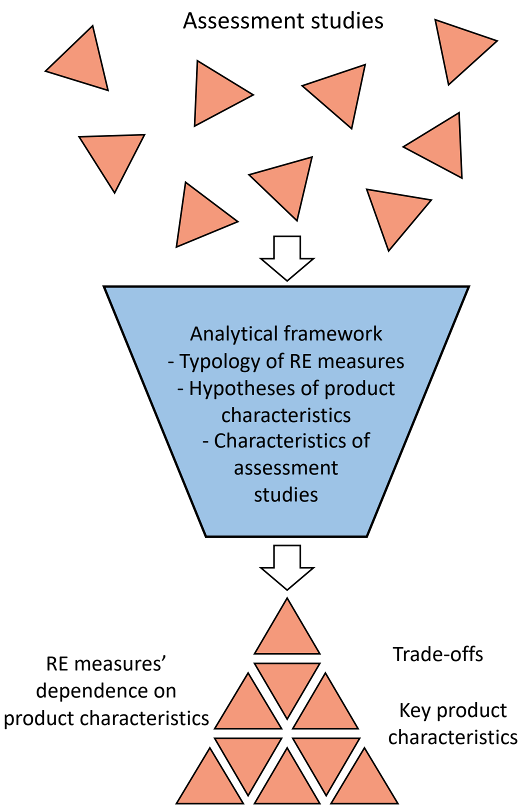


*Figure 5: Schematic representation of the method used to synthesise environmental assessments, including collection of assessment studies from literature, analysis and categorisation using an analytical framework and finally synthesis to identify patterns, trade-offs and key product characteristics (Paper I).*

Both scientific and grey literature were searched for environmental assessments (mainly including variants of LCA and MFA) that compare a product or service before and after implementing some resource efficiency measure. Studies conducted by companies and/or academic partners within the Mistra REES research programme were also included (Mistra REES, 2020). The selection criteria were that each study should compare at least one measure to some reference, and that the study be transparently presented regarding key assumptions, data, methods and results. Studies were selected by emphasising resource efficiency measures (section 2.1) for efficient, effective and extended use. This was complemented with a selection of cleaner production and recycling measures.

Each of the collected assessment studies was subsequently analysed and categorised, first according to the typology of measures presented in section 2.1. Further analysis and categorisation were done, by listing potentially relevant product characteristics that were hypothesised to be relevant for the outcome of different measures. Each product characteristic was then tested to find patterns between product characteristics, measures and their resource efficiency outcomes in each case. This synthesis led to the identification of a number of key product characteristics of particular relevance.

To achieve this, the results of each study were noted in terms of improvements or deteriorations in resource efficiency (in the categories material efficiency, energy efficiency and environmental performance). This was done semi-quantitatively, where significant improvements or deteriorations compared to the reference were denoted ‘+’ or ‘-’, respectively, while minor changes (less than 2.5%) were denoted ‘0’.

All collected information was gathered in a database which allowed systematic sorting and analysis on various levels and dimensions. A systematic mapping of measures, product characteristics and resource efficiency outcomes enabled pattern-identification of characteristics that correlated (positively or negatively) with resource efficiency improvements in various cases.

In practice, the entire process was iterative. The analysis and synthesis of assessment studies prompted further developments of the framework which in turn caused the reclassification of studies and collection of additional assessment studies.

# Chapter 4 Synthesis of assessment studies

The following sections will address the research question formulated in section 1.1, starting with the first, on achieving more empirically grounded guidelines for resource efficiency (RQ1).

The procedure described in section 3.3 and summarised in Figure 5 was employed in Paper I to collect, categorise and synthesise assessment studies of resource efficiency measures from literature. The aim was to guide the implementation of such measures, by investigating which product characteristics are relevant for resource efficiency (RE) outcomes (out of a list of characteristics hypothesised to be of relevance). 59 assessment studies were collected, covering 124 measures applied in various contexts, covering all resource efficiency measures (except some post-use measures) and many different types of products across many sectors.

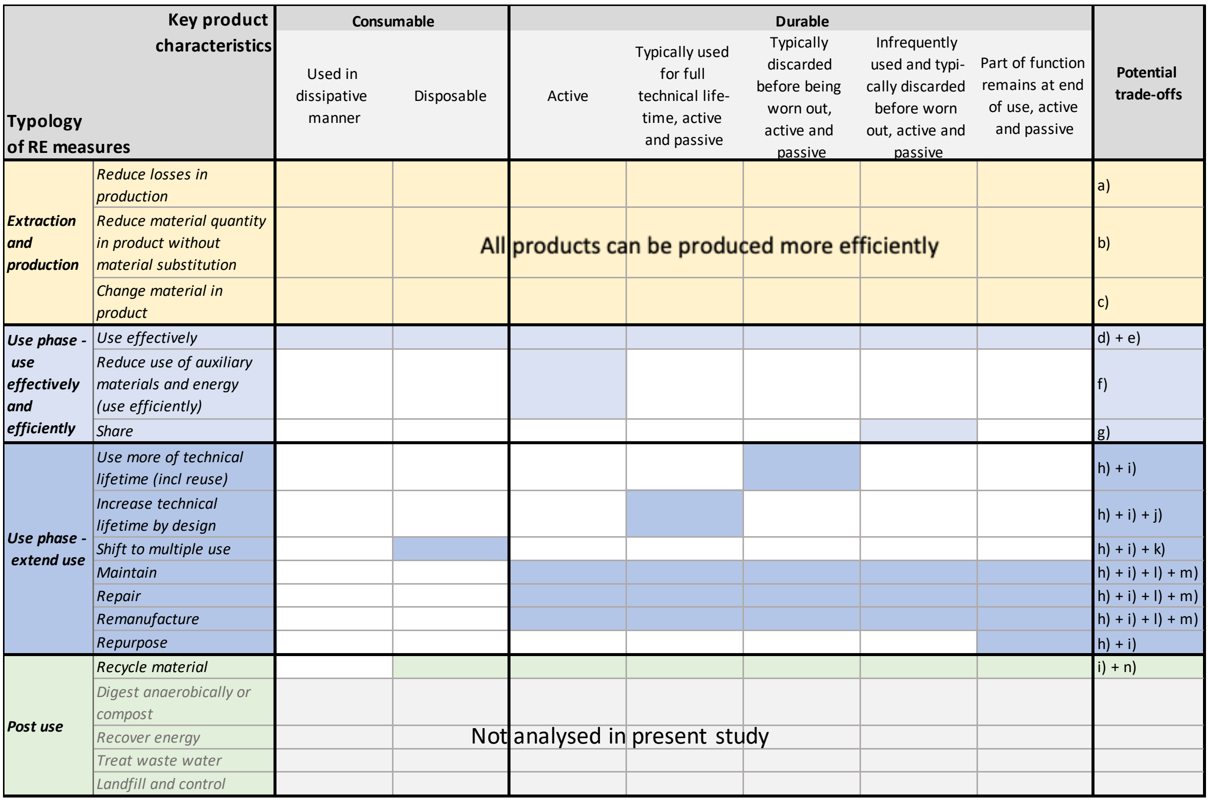
Categorisation, systematic mapping and synthesis of the collected assessments enabled the identification of product characteristics that were of particular importance for the outcome of each resource efficiency measure. The product-based guidelines presented in Figure 6 were formulated based on this synthesis. It shows what resource efficiency measures tend to generate environmental improvements for which types of products and also indicates related trade-offs.

A key distinction was made between *durable* and *consumable* products. Much of the discussion about circular economy concerns durable products and extension of their use, while consumable products, for which use life extension is not relevant, are given less attention. Rather than extending use life, consumables can be produced more efficiently and used more effectively. A further distinction can be made between consumables that are *disposable* or used in a *dissipative* manner. A dissipative product (e.g. food and detergents) is transformed during use, after which it becomes intangible or dissipated. By contrast, a disposable product is typically used once, after which it is disposed of while still existing as a distinct object (e.g. packaging, that can be material recycled or redesigned for multiple use). A dissipative product cannot be recycled, thus it is particularly important to produce it more efficiently and use it more effectively.

For *durable* products all measures aiming for extended use such as maintain, repair and reuse are relevant, in addition to efficiency in production and post-use. Further, aspects pertaining to the use phase are important in determining what measures are effective. For *active* products, use-phase efficiency (reduced use of energy or auxiliary material) is important and may even outweigh the benefits of extending the use of the product. For *infrequently used products*, sharing is a potentially suitable measure, although it does not on its own improve resource efficiency for products that already tend to be used for their full technical lifetime. Repurposing is suitable for *products with remaining functionality at the end of use*.

The collected studies did not allow for systematic testing of the importance of product complexity and related possibility to disassemble. However, *product complexity* is often discussed as a key characteristic for the efficacy of restorative measures and recycling, e.g., in Ljunggren Söderman and André (2019) and in eco-design literature (Ceschin and Gaziulusoy, 2016; Luttropp and Brohammer, 2014; Sundin, 2009). Similarly, the importance of hazardous or scarce materials could not be tested, although there were examples regarding substitution of scarce metals (Arvidsson et al., 2016; Reuter, 2016).

Generally, which post-use measures are suitable depends on material characteristics rather than on the product characteristics mentioned above. Similarly, measures for resource efficiency in extraction and production are applicable to all products, regardless of product characteristics.



| Typology of RE measures | Key product characteristics | Consumable | | Durable | | | | | Potential trade-offs |
|---|---|---|---|---|---|---|---|---|---|
| | | Used in dissipative manner | Disposable | Active | Typically used for full technical life-time, active and passive | Typically discarded before being worn out, active and passive | Infrequently used and typically discarded before worn out, active and passive | Part of function remains at end of use, active and passive | |
| ***Extraction and production*** | *Reduce losses in production* | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | a) |
| | *Reduce material quantity in product without material substitution* | All products can be produced more efficiently | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | b) |
| | *Change material in product* | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | c) |
| ***Use phase - use effectively and efficiently*** | *Use effectively* | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | d) + e) |
| | *Reduce use of auxiliary materials and energy (use efficiently)* | | | ▨ | | | | | f) |
| | *Share* | | | | | | ▨ | | g) |
| ***Use phase - extend use*** | *Use more of technical lifetime (incl reuse)* | | | | | ▨ | | | h) + i) |
| | *Increase technical lifetime by design* | | | | ▨ | | | | h) + i) + j) |
| | *Shift to multiple use* | | ▨ | | | | | | h) + i) + k) |
| | *Maintain* | | | ▨ | ▨ | ▨ | ▨ | ▨ | h) + i) + l) + m) |
| | *Repair* | | | ▨ | ▨ | ▨ | ▨ | ▨ | h) + i) + l) + m) |
| | *Remanufacture* | | | ▨ | ▨ | ▨ | ▨ | ▨ | h) + i) + l) + m) |
| | *Repurpose* | | | | | | | ▨ | h) + i) |
| ***Post use*** | *Recycle material* | | ▨ | ▨ | ▨ | ▨ | ▨ | ▨ | i) + n) |
| | *Digest anaerobically or compost* | | | | | | | | |
| | *Recover energy* | Not analysed in present study | | | | | | | |
| | *Treat waste water* | | | | | | | | |
| | *Landfill and control* | | | | | | | | |

a) Reduced production losses <=> energy use for avoiding losses
b) Risk of losing function, e.g. durability
c) Risk of burden shifting when substituting materials
d) No identified trade-offs, except: chemicals with higher functionality vs risk of more hazardous constituents
e) No identified trade-offs, except: reduced use phase impact <=> production of sensors (when required)
f) Reduced use-phase impacts <=> Increased production impacts
g) Sharing can increase car transportation for users accessing the shared stock
h) Use-phase efficiency <=> benefits of use extension (for active products with technological development towards use-phase efficiency)
i) Risk of keeping hazardous substances in circulation
j) Durability <=> Amount (or impact) of materials
k) Benefits of multiple use <=> increased impact from production and maintenance
l) Maintenance can increase transportation
m) Design for disassembly can increase material use
n) Impacts from recycling need to be smaller than impacts from primary production

*Figure 6: The product characteristics for which different resource efficiency measures are suitable (coloured tiles in the centre of the figure), as well as potential associated trade-offs (indexed alphabetically to the right), from Paper I.*

Some products can be characterised at a system level, such as to which sector it belongs to or which life cycle phase tends to dominate environmental impacts. Of these system-level characteristics, the synthesis revealed the *dominating life cycle phase* to be of key importance for which resource efficiency measure is effective. This relates to the already discussed active products, for which use-phase efficiency is important. For products for which the extraction and production of raw materials dominate, avoiding losses throughout the life cycle becomes important. No correlation between type of industry and the suitability of measures could be found in the collected material. Finally, the *pace of development* plays an important role in trade-offs for active products for which use-phase efficiency is improved over time.

An important limitation of the study was that the synthesis only concerned studies of physical measures for resource efficiency, as presented in section 2.1 (for more detailed limitations, see Paper I). However, while the guidelines are not aimed at any particular application, the physical measures can be implemented in a variety of ways. This includes product design or the adoption of new policies or business models (see Chapter 6, which addresses the potential environmental consequences of adopting new business models). In fact, any designer, consumer, business or policy-maker involved in decision-making around a specific type of product can use the guidelines in Figure 6. This will allow them to find what resource efficiency measures are suitable and what trade-offs to be aware of depending on the characteristics of the products involved. That being said, Willskytt and Brambila-Macias (2020) developed the guidelines further into a tool to be used for eco-design. Similarly, the guidelines can be further refined and developed in future research, to be more easily accessible for other applications.

# Chapter 5 Prospective assessment of metal 3D printing

The second research question (RQ2) concerns the environmental improvements that can be achieved by following guidelines and heuristics. In Paper II a prospective LCA was carried out scrutinising the environmental expectations of 3D printing (also called additive manufacturing), specifically of powder bed fusion (PBF) applied in the automotive industry.

3D printing is an umbrella term for several techniques used to construct three-dimensional objects by binding material together until a desired shape and size is achieved based on 3D model data (ASTM International, 2012; Rombouts et al., 2006). Figure 7 shows a schematic overview of the technology in the case of metal 3D printing by PBF. In the PBF process, a thin layer of feedstock material in powder form is placed in a chamber and a laser melts selected parts. Then another layer of powder is spread on top and again selectively melted, thus fusing with the layer beneath. This process is repeated until a solid structure is achieved, according to the digital specifications (Louvis et al., 2011). The thickness of every layer (ca 20–40 μm) depends on the powder and machine specifications and settings, and in turn affects the resulting surface quality and the need for post-processing (Dawes et al., 2015). An additional factor, affecting both post-processing needs and total energy consumption, is the orientation of the part being printed (Mognol et al., 2006). Support structures have to be printed to keep the part stable until the print is complete, after which they can be removed. The need for support structures can be minimised by careful part orientation (Faludi et al., 2017).

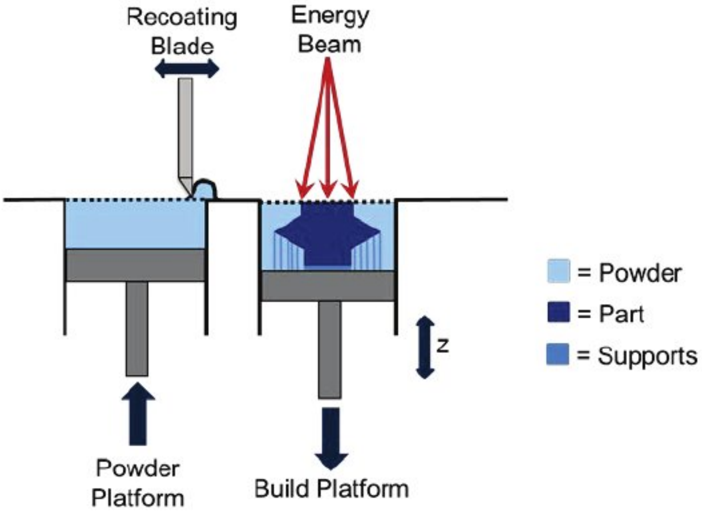


*Figure 7: Schematic of the powder bed fusion process. Reused here with permission (Moylan et al., 2014).*

Several metallic materials are available as feedstocks; for example, powders made from aluminium alloys, steel alloys, nickel alloys and titanium alloys (Wohler's Associates, 2016). The method for producing these powders is called atomisation and involves a pressurised gas, liquid or plasma being shot at molten metal falling in a chamber. This metal breaks it into droplets that solidify into spheroids on their way down (Dawes et al., 2015; Yule and Dunkley, 1994).

3D printing is an emerging technology in the early stages of adoption and is believed to have revolutionising potential for many industries. Metal 3D printing by laser melting is a version that has attracted significant interest (Walachowicz et al., 2017; Wohler's Associates, 2016). The automotive industry in particular has expressed expectations that metal 3D printing can improve resource efficiency by allowing redesigns that reduce vehicle weight and thus fuel consumption (Lifset, 2017; Volvo Group, 2017). Other potential advantages include on-demand spare part printing as well as redesigns for fewer components and additional functionality (Ford and Despeisse, 2016; Kellens et al., 2017; Priarone et al., 2017). The disadvantages of 3D printing include a slow and energy-consuming printing process and limitations on what materials and sizes can be printed (Gutowski et al., 2017).

The aim of the case study in Paper II was to assess the environmental effects of these potential advantages and disadvantages. The study was a collaboration with Volvo Group, who provided a large part of the input data. To investigate the potential future environmental effects of 3D printing, prospective LCA methodology was applied to the case of truck engines, whose metal components were redesigned to be 3D printed. The resulting environmental impacts were calculated from a life cycle perspective. The redesign was carried out by Volvo Group in an internal project and was interpreted for the purposes of this LCA study with the aid of Volvo experts.

As mentioned in section 3.1.1, three scenarios were formulated. The reference scenario, S0, represented conventional manufacturing of the engine. This was compared to two 3D printing scenarios. Scenario S1 represents the present state of metal 3D printing technology, with limitations on the size of components that can be printed, meaning that only smaller components can be redesigned for 3D printing and thus only a small weight decrease. Furthermore, there is a limited selection of materials that can be used for printing, including stainless steel, nickel alloys and titanium alloys. Low-alloy steel, for example, cannot be used for printing. Scenario S2 was placed a decade into the future, when 3D printing can be expected to have matured and spread significantly. In scenario S2 it was assumed that large components can also be printed and consequently a larger share of components can be redesigned for 3D printing and lower weight. Furthermore, options for printing with low-alloy steel instead of nickel alloys or high-alloy steel were assumed to be available. This is important since low-alloy materials have lower environmental burdens. The engine weight in the reference scenario was 533 kg. The two versions of the redesigned engine were 499 kg and 418 kg for S1 and S2 respectively. The functional unit serving as the basis of comparison was set to 2.55 Mton·km, representing the function of one engine that enables the transportation of 8 500 kg load over 300 000 km (an approximate average capacity and lifetime of a light distribution truck).

A life cycle model was constructed, and data were gathered into a life cycle inventory. Several weighting methods were applied to identify the types of impact that were the most significant (see section 3.1 as well as Paper II for details). Greenhouse gas emissions were identified as one key type of impact in this case. Figure 8 shows the emissions of greenhouse gases (in CO2-equivalents) per the life cycle of a truck, i.e., 2.55 Mton·km. The energy-intensive 3D printing process causes the emissions from engine production to increase compared to the reference scenario. Conversely, emissions from fuel production and consumption are reduced due to the lower weight of the vehicle with a 3D printed engine. In scenario S1, the net result is similar to that of the reference scenario, which indicates that it is not favourable to implement 3D printing of engines at the present. In the future scenario, S2, the net results show an improvement of approximately 15%, implying that 3D printing could be favourable. However, a key assumption underlying this result is the use of low-fossil electricity for the printing process.

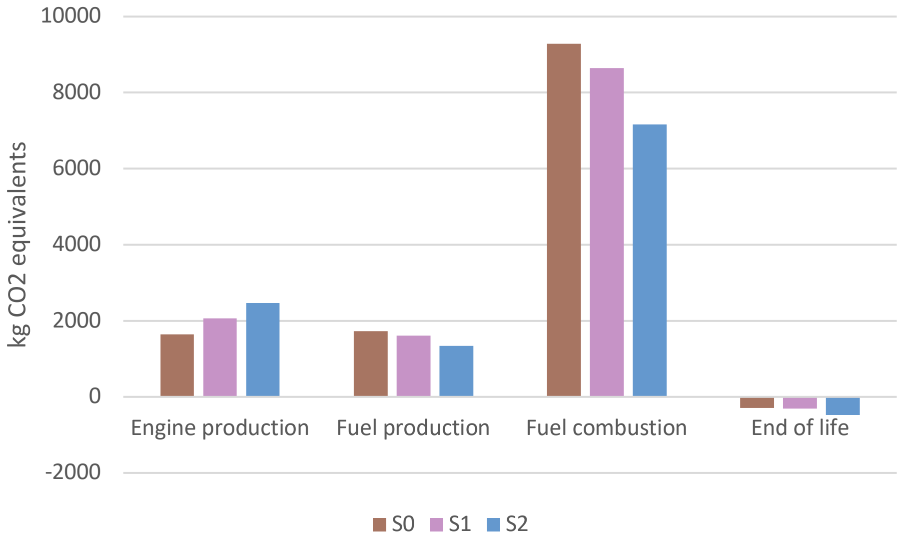


*Figure 8: Emissions of greenhouse gases for the different life cycle stages of the engine, as represented by kg $CO_2$-equivalents per 2.55 Mton·km, from Paper II.*

Material resource use was another key impact according to the applied weighting methods. Metals presently available for 3D printing are highly alloyed (including stainless steel, nickel and titanium). Consequently, when redesigning components such as the connecting rod to be 3D printed, with current printing technology, there has to be some material substitution. A connecting rod is typically manufactured from low-alloy steel, but in order to be 3D printed the material must be changed to stainless steel for example. As shown in Table 1, this redesign for 3D printing will reduce the component's weight but will sharply increase the environmental impacts from resource consumption. This example illustrates why overall material resource use is increased in scenario S1, namely due to substitution of iron and low-alloy steel with high-alloyed steel. In the future scenario S2, a technology development allowing printing with materials of a lower impact (such as low-alloy steel) was assumed. Therefore, the sharp increase in impacts from material resources can be avoided if such a development is realised.

*Table 1: Weight of an example component (connecting rod) in the different scenarios, along with the weighted environmental impacts expressed in environmental load units (ELU), according to EPS endpoint weighting (Steen, 2015).*

| Connecting rod | | | |
|---|---|---|---|
| | Material | Weight [kg] | EPS impacts [ELU] |
| S0 | Low-alloy steel | 11.1 | 18.6 |
| S1 | Stainless steel (PBF) | 8.3 | 729 |
| S2 | Low-alloy steel (PBF) | 8.3 | 20.7 |

Detailed interpretation, including testing of the robustness of the results, can be found in Paper II. However, a limitation of the study, in addition to the inherent uncertainties of prospective LCA, is that the slow speed of the 3D printing process is not accounted for. This will influence the potential implementation of the technology in real manufacturing. Another factor was the post-processing, which may require significant efforts and energy but was only modelled very simply as a minor material loss. There are also additional potential benefits from the technology that were not accounted for in the model. For example, developments in design for added functionality could allow for further reduced fuel consumption, spare part printing could allow for improved repairs, and streamlined supply-chains could allow for improved logistics.

In summary, 3D printing applied in the automotive industry has the potential to reduce environmental impacts in the future by reducing weight to decrease fuel consumption. However, this is only true with low-fossil electricity for printing and with technology development towards the possibility of printing materials with lower environmental impact such as low-alloy steel.

# Chapter 6 Business model LCA

The previous chapter addressed the second research question (RQ2) in the case of 3D printing, and here will be scrutinised the expectations placed on alternative business models to achieve decoupling of economic performance from environmental impact. These alternative business models include, e.g., SBM, CBM and PSS. They are expected to make money-making less dependent on the continuous throughput of material by putting a larger emphasis on services, by valorising waste, and by reusing resources, thus enabling companies to achieve decoupling (Bocken et al., 2016; Schaltegger et al., 2012). To test whether such business models actually enable decoupling in real cases, we first need to be able to assess business models in a way that takes into account both economic and environmental dimensions (as explained in section 3.1.2). A method developed for this purpose is presented in section 6.1. The method is then applied in section 6.2 on a case comparing renting and selling of jackets in the Swedish apparel sector, in order to test whether the expectations on CBMs, PSS and SBMs are fulfilled in such a case.

## 6.1. A new method for environmental assessment of business models

Paper III presents a new method, named business model LCA (BM-LCA), developed according to the principles described in sections 3.1.2 and the procedure in section 3.2. The method differs from mainstream LCA on several key points, the main one being that it takes business models themselves as the unit of analysis. The function of a business model is interpreted as generating monetary value, why the functional unit is expressed in terms of profit. Since the widespread adoption of the business model concept, the primary purpose of its use has been to support profit generation (Bocken et al., 2014; Magretta, 2002). There are arguments for an expanded view of the purpose of businesses (Geissdoerfer et al., 2018b), but from the perspective of the business and its owners and shareholders the purpose might indeed be sustained economic performance, why this is chosen here as the core purpose of a business. While economic performance can be measured with various indicators, including profit margin and rate of return, a simple approach was chosen for BM-LCA, using profits (revenues minus costs) as the functional unit.

All phases and detailed steps of BM-LCA are summarised in Table 2. In short, the Goal and Scope phase in LCA has been elaborated and divided into two parts. The first phase describes the key features of each business model under consideration as well as the related product system, including how the amount of production depends on the number of customer transactions. The second phase defines a profit-based functional unit, thereby establishing a quantitative basis for comparison between business models. Equations are then set up to couple monetary flows and physical flows. The number of transactions required in each business model is then calculated and the associated amount of production is derived. Finally, standard

LCA procedure is followed for life cycle inventory (LCI) analysis, life cycle impact assessment (LCIA) and interpretation of results.

In sections 6.1.1 and 6.1.2 each phase of BM-LCA is detailed, for a generic business comparing two business models, although the method allows for any number of models to be compared.

*Table 2: A summary of BM-LCA, along with detailed steps for each phase and a brief description of how to carry out each step (from Paper III).*

**Business model LCA**

| Phase | Description of each step |
|---|---|
| Goal and scope: descriptive phase | Give general description of the setup of each business model to be compared and of the related product(s) and state the time period to consider. |
| | Define system boundaries and environmental impact categories of the assessment. Map actors in the product chain. |
| | Find the connection of how the amount of production, $q$, depends on the number of transactions, $t$, for each business model. |
| Goal and scope: coupling phase | Step 1: Define the functional unit as the profit, $\pi$, that each business model must achieve. |
| | Step 2: Identify all of the business' costs and revenues associated with running one of the business models for the stated period. Find conversion factors, $f$, to couple costs and revenues to customer transactions, $t$. Set up an equation for the profit as revenues minus costs: $\pi = f_{revenue} * t - f_{direct} * t - f_{indirect} * t - f_{contingent} * t$ |
| | Step 3: Solve the equation to find the transactions, $t$, required to reach the profit. Derive the required amount of production, $q$. |
| | Step 4: Repeat steps 2 and 3 for every business model to be compared. |
| Life cycle inventory | Construct a system model and quantify all environmentally relevant flows, scaled according to the functional unit. |
| Life cycle impact assessment | Aggregate all flows from LCI and quantify their effects on the chosen environmental impact categories. |
| Interpretation | Analyse the results and scrutinise their robustness to identify pros and cons of compared business models. |

### 6.1.1. Goal and scope: descriptive phase

The goal and scope definition involves defining the purpose of the study, the business models considered, environmental impact categories and system boundaries. The purpose is thus to assess and compare the environmental effects of at least two different business models. The system boundaries should at least cover the life cycle of the products involved from cradle to grave. The time period, geographic limitations and environmental impact categories are defined according to the case in question. Data sources and quality should reflect the real situation of the business to the largest extent possible, particularly as regards economic data.

The business models under consideration are described in terms of the type of customer transactions that take place, whether the business retains or sells ownership of the products, and how product stocks (if any) are maintained. The product(s) associated with the business models must also be defined and described in terms of their most relevant characteristics (such as function, lifetime, and material composition).

Furthermore, a connection must be established regarding how the amount of production, q, depends on the number of transactions, t. This can be done by applying a PCO approach, which involves mapping the actors in the product chain to find the life cycle steps belonging to the business, suppliers and clients in order to identify which transactions take place and what exchange of goods and material are associated with each transaction. For instance, in a linear sales model every customer transaction implies the sale of a product (which first has to be produced or acquired from a supplier), and consequently t and q will be equal. In a rental model, however, q will depend on the rate at which products are worn out and replaced, which in turn depends on the number of rental transactions. For a pure service model, there is no exchange of goods or materials between business and customer. However, even pure services usually depend on material flows: hair salons require premises, shampoo and water, and IT services require physical networks, servers and electricity.

### 6.1.2. Goal and scope: coupling phase

The next phase follows the procedure in Figure 9. It starts by defining a functional unit that will serve as the basis of comparison between the business models. This will then allow for setting up equations that couple the material and monetary flows in the business model, which will give the number of transactions and associated production to reach the defined profit level. This process is then repeated for each business model to be compared.

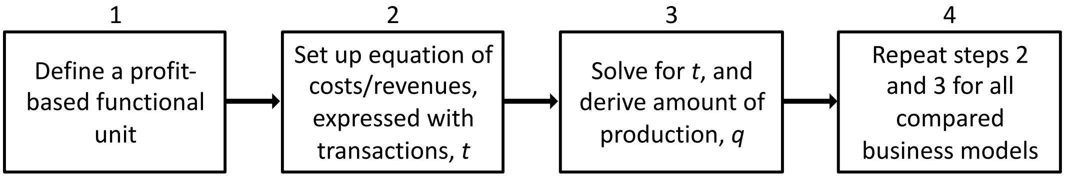


*Figure 9: Procedure for coupling monetary and physical flows and finding the number of customer transactions and associated amount of production.*

In more detail, a functional unit is defined in step 1. As established, the function of a business model is interpreted as its economic performance, which should be equal for each compared model. Consequently, the functional unit is defined as the following:

- A certain amount of profit, π, over a business period, T, from customer transactions for a particular set of products from a particular business

The profit level chosen may be based on either the stated goals of the business, if the aim is to support business model innovation, or average historical profits, if the aim is to assess current business models.

In step 2 an equation is set up to couple the monetary and material flows for each of the compared business models in order to find the number of customer transactions, t. This in turn determines the necessary number of products, q. Start by finding all revenues and costs related to operating the business model during the period T depending on the transactions, t. Throughout this paper we will use a generic cost structure adapted from Norris (2001) to represent different types of costs:[3]

- Direct costs (e.g. cost of production, labour, capital investment and waste disposal)
- Indirect costs (costs that cannot be allocated directly to a product or process, e.g. administrative overhead costs)
- Contingent costs (e.g. fines, penalties and liabilities)

The cost categories chosen should be relevant to the business under consideration. The choice also depends on whether the analysis will be static (disregarding the time-value of money) or dynamic (in which case, e.g., interest rates and discounted future costs can be taken into account). Importantly, only costs carried by the business itself should be included (i.e. not external or customer costs).

An equation can be set up for the profit ($\pi$) as the revenues ($R$) minus the costs ($C$) of the business model:

*Equation 1:*

$$\pi = R - C_{direct} - C_{indirect} - C_{contingent}$$

In order to solve this equation for the number of transactions, t, the revenues and costs must be expressed in terms of $t$. For this purpose, we introduce a coupling factor, $f$, that allows revenues and costs to be written as in Table 3. The factors will be different for each cost or revenue and for each business context, and will couple the money flows to the customer transactions in each case.

---

[3] Note that intangible and external costs are excluded as they do not directly determine the costs and revenues of a business.

*Table 3: Revenues and costs in the first business model, expressed in terms of the number of customer transactions*

| Revenue or cost | Expression in terms of number of transactions, t |
|---|---|
| $R =$ | $f_{revenue} * t$ |
| $C_{direct} =$ | $f_{direct} * t$ |
| $C_{indirect} =$ | $f_{indirect} * t$ |
| $C_{contingent} =$ | $f_{contingent} * t$ |

Equation 1 can now be written in terms of transactions and coupling factors:

*Equation 2:*

$$\pi = f_{revenue} * t - f_{direct} * t - f_{indirect} * t - f_{contingent} * t$$

A coupling factor must be found for each cost and revenue. To illustrate what the factors $f$ could be, $f_{revenue}$ could for example depend on the price of one transaction, so the revenues would be:

$$R = transaction\,price * t$$

Hence, the coupling factor, $f_{revenue}$, in this example is equal to the transaction price.

A less straightforward example involves connecting indirect costs to the number of transactions. Indirect costs are often semi-fixed, such as the costs for office space, which are not directly dependent on transaction or production volumes. This needs to be solved for each individual case, but one example could be to express how the indirect costs depend on the amount of real estate that the business uses. Indirect costs would then be:

$$C_{indirect} = indirect\ cost\ per\ unit\ of\ real\ estate * amount\ of\ real\ estate$$

We can then estimate how much real estate is needed to sustain a certain number of transactions during a specific period. Then the amount of real estate can be expressed as follows:

$$amount\ of\ real\ estate = \frac{t}{\#transactions\ sustained\ per\ unit\ of\ real\ estate}$$

The indirect costs can now be written as:

$$C_{indirect} = \frac{indirect\ cost\ per\ unit\ of\ real\ estate * t}{\#transactions\ sustained\ per\ unit\ of\ real\ estate}$$

Hence, in this example, the coupling factor is:

$$f_{indirect} = \frac{indirect\ cost\ per\ unit\ of\ real\ estate}{\#transactions\ sustained\ per\ unit\ of\ real\ estate}$$

In other cases, indirect costs will be truly independent of customer transactions, there is no coupling factor. The corresponding cost in Table 3 will then be a fixed number, independent of transactions.

In step 3, the known profit, $\pi$, and coupling factors can be used to solve Equation 2 for the transactions, t, required to reach the profit in the functional unit:

*Equation 3:*

$$t = \frac{\pi}{f_{revenue} - f_{direct} - f_{indirect} - f_{contingent}}$$

Once t is calculated, the required amount of production, q, associated with that level of transactions can be derived based on the connection between the t and q established in the descriptive phase.

Lastly, steps 2 and 3 are repeated for each of the business models to be compared. In other words, set up an equation, solve it for t by finding the corresponding coupling factors, and finally find the necessary amount of production, q, depending on the business model in question.

Armed with the number of customer transactions and number of associated products needed to reach the same profit in each business model, it is now possible to feed these types of parameters into the LCA and calculate the environmental impacts. This is done by applying conventional LCA methodology for building an LCI, carrying out LCIA and interpreting the results.

## 6.2. Comparing renting and selling in the Swedish apparel sector

BM-LCA will be applied here, partly to investigate a case comparing renting and selling of jackets by a Swedish apparel company, and partly as an illustration with detailed steps of how BM-LCA can be applied in practice.

### 6.2.1. Goal and scope: descriptive phase

Here will be described the objective of the assessment, the sales and rental business models and the related product system. Details on data collection and sources and the choice of impact categories for the environmental assessment can be found in Paper IV.

The objective of the assessment was to compare the case company's sales business model with a rental business model for polyester jackets by answering the following specific questions:

1. Can a rental business model for jackets reduce environmental impacts while maintaining profitability compared to the sales business model?
2. What are the environmental hotspots in the rental and sales models?
3. Is there any burden-shifting between types of impact or different parts of the life cycle?
4. What are the most significant parameters affecting the performance of the rental business models?

The sales model implies that every garment produced is sold to a customer at an established price. Consequently, the number of transactions during a certain period equals the number of garments that need to be produced. The company also offers customers a free repair service. In the rental model the company retains ownership of products while customers pay a price to access them one day at a time. The company maintains the garments, including laundry between customers and repairs. When garments are deemed too worn from repeated rentals, they are sold second hand at a reduced price. In both business models, the company accepts old jackets in return from customers for recycling.

The investigated jacket is made of polyester, with the same design in both business models. It is composed of: (i) an outer face fabric (with a fluorocarbon-free water repellent), (ii) an interior backing fabric and (iii) laminated to the face fabric, an intermediate waterproof membrane that enables humidity to escape from the wearer. The face fabric is made of recycled polyester while the backing, membrane and zipper are made of virgin polyester.

The jacket life cycle in the sales and rental business models is visualised in Figure 10, including the technical system (described in detail in Paper IV) as well as indications of the monetary flows connected to interactions between the actors involved that generate costs or revenues for the case company. Coloured boxes represent a different actor responsible for that process, red for external suppliers (in Japan and Estonia), blue for the case company and orange for customers. Not depicted in the figure are background systems like electricity and water production, which are modelled according to the location where each process takes place.

Figure 10 also indicates that the revenues and costs from the company's perspective are taken into account. A simple cost structure of direct and indirect costs was considered, where direct costs depend on the volume of production, while indirect costs were represented by fixed costs. Revenue streams were divided into input-based revenues, generated when customers receive the ownership of a product, and usage-based revenues, generated when customers rent a product.

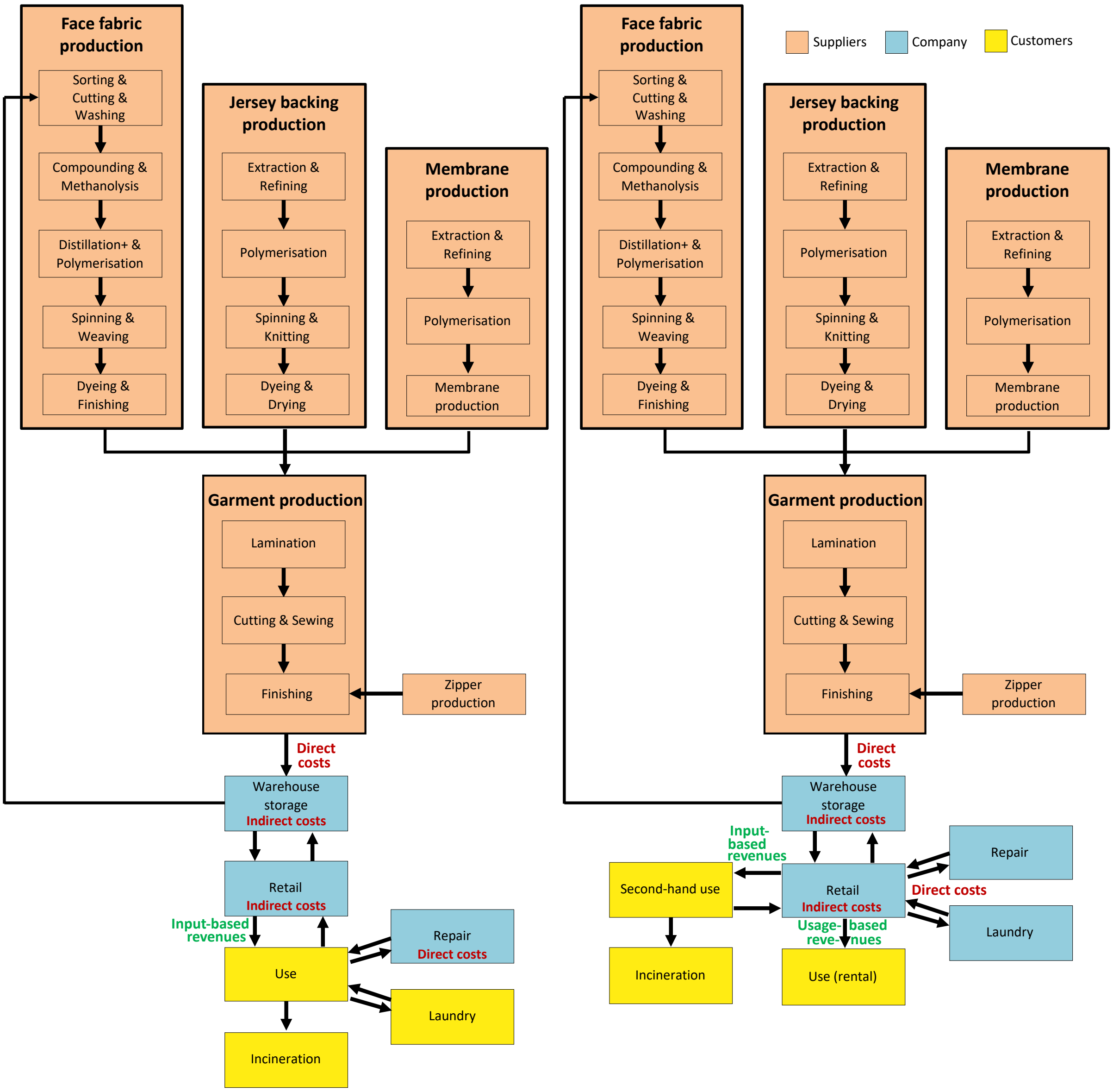


*Figure 10: The socio-technical product system, representing a sales business model (left) and a rental business model (right) for jackets. Arrows represent material flows and colours represent different actors. Costs and revenues for the case company are indicated by red and green text, respectively. Some costs or revenues are associated with running a process like the warehouse, others stem from the exchange of material to/from another actor. In the latter case, they are indicated next to the corresponding material flows*

### 6.2.2. Goal and scope: coupling phase

In the coupling phase is modelled the relationships between monetary and material flows, starting from the profit to then derive the required number of transactions and volume of production to achieve that profit in each business model. The first step was to define a profit-based functional unit, here 'a certain amount of profit, π, over a business period of 30 days, from the transactions of the studied jackets'. For the sales model, the profit generated by the jackets was estimated using company data on sales volume, price and costs, combined with literature data on common profit margins in the industry. The number of transactions and consequent production required to reach the same profit in the rental model were then calculated.

To achieve this, all types of money flows and economic parameters were defined and calculated or estimated. In Table 4, the adopted cost structure (for both business model, in the name of simplicity) is presented.

*Table 4: Cost and revenue categories and their assigned symbol and description. Purple is only relevant for the sales business model, blue only for the rental business model.*

| **Monetary flows categories** | **Symbols** | **Descriptions** |
|---|---|---|
| Input-based revenues | $RE_s$ | Total revenues generated by costumers paying for a new jacket in order to obtain the ownership |
| | $RE_{r,2nd}$ | Total revenues generated by costumers paying for a second-hand jacket in order to obtain the ownership |
| Usage-based revenues | $RE_r$ | Total revenues generated by customers paying for the use of a jacket |
| Production costs | $C_{prod}$ | Total aggregated cost that includes the production of textile fibres, the manufacturing and the transportation[4] costs. |
| Distribution costs | $C_{distr}$ | Total cost for distributing jackets from the central warehouse to the company stores |
| Laundry costs | $C_{laundry}$ | Total cost for washing jackets |
| Repair costs | $C_{repair}$ | Total cost for repairing a jacket in case of damage |
| End-of-Life costs | $C_{EoL}$ | Total cost for the transportation of collected jackets to the chemical recycler supplier in order to recover material for the fibre production for new face fabric |
| Employee costs | $C_{emp}$ | Total cost incurred to pay employees that operate the stores and cover social fees |
| Overhead costs[5] | $C_{OH}$ | Total cost for recurring expenses, e.g., rent, utilities and storage |

The numerical economic data is summarised in Table 5. Notable parameters include the 'rental efficiency' ($E_r$), which describes what share of garments in the rental stock are rented by customers at any given time. It depends on time required for maintenance activities and what over-capacity of the stock is needed to meet fluctuating demand. Another key parameter is the rental lifetime (*RL*) which is how many use days a jacket can provide before being too worn out and removed from the rental stock. The removed jackets are sold second-hand at 60% of the original price and the rental stock is replenished by adding a newly produced jacket.

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[4] Refers to transport between the external suppliers and the warehouse of the company.

[5] Overhead costs are considered to be semi-fixed, and independent of sales volume, until a level is reached where, e.g., a new store has to be opened.

*Table 5: Values of costs, prices and parameters to calculate total costs and revenues and related physical flows. * indicates parameters valid both in the sales and in the rental business model.*

| Symbol | Description | Values | Sources |
|---|---|---|---|
| $k_{prod}$* | Unit cost of production per jacket | 2500 SEK/jacket | Derived from the sales and an estimated mark-up margin of 50% |
| $k_{distr}$* | Unit cost of distribution per jacket | 0.14 SEK/jacket | Estimated by referring to Maibach et al. (2006) and by considering the average distance of the stores from the warehouse (approximately 449 km) |
| $K_{laundry}$ | Unit cost of laundry per transaction | 70 SEK/transaction | Provided by the company |
| $k_{repair}$* | Unit cost of repair per transaction | 8 SEK/transaction | Provided by the company |
| $k_{EoL}$* | Unit cost of recycling per jacket | 18 SEK/jacket | Estimated by summing the distribution costs and the cost of shipping calculated on worldfreightrates.com by considering the distance between the warehouse and the external supplier |
| $k_{emp}$* | Unit cost per employee | 39 300 SEK/ employee | Estimated by considering an average salary of a shop assistant in Sweden (26200 SEK/month) and adding social costs, estimated at 50% of the salary costs (Business Sweden, 2020) |
| $k_{OH}$* | Unit cost per store | 5000 SEK/store | Provided by the company |
| $P_s$ | Price for buying a jacket | 5000 SEK/jacket | Provided by the company |
| $P_r$ | Price for renting a jacket | 600 SEK/rent | Provided by the company |
| $P_{2nd}$ | Price for buying a second-hand jacket | 3000 SEK/jacket | Provided by the company |
| $N_s$ | Number of stores | 4 stores | Provided by the company |
| $SS$* | Storage capacity | 50 jackets | Provided by the company |
| $EPS$ | Number of employees per store | 1 employee | Assumed |
| $RL$ | Rental lifetime | 200 use days | Provided by the company |
| $E_r$ | Rental efficiency | 60 % | Provided by the company |
| $U_r$ | Average use days per rental transaction | 5 use days | Provided by the company |
| $CR$ | Collection rate | 50 % | Assumed |
| $T$* | Business period | 30 days | Established |

The connection between monetary and physical flows was done via the functional unit, defined as the monthly profit from the studied jackets. The functional unit was quantified based on the collected economic data for the sales model, together with the monthly sales volume, estimated at 200 transactions per month ($t_s = 200$ transactions). As shown in Table 6 the monthly profit, $\pi_s$, amounts to 319 391 SEK. This translates to a physical flow 200 jackets per month, since in the linear model the number of sold jackets equals the required production ($q_s = 200$ jackets).

*Table 6: Monthly revenues and costs in the sales model (30 is the conversion factor between months and days).*

| **Revenue or cost category** | **Connection in equation form** | **Calculated revenues and costs (SEK)** |
|---|---|---|
| Revenues from sales transactions | $RE_s = P_s*t_s$ | 1 000 000 |
| Production costs | $C_{prod} = k_{prod}*q_s$ | 500 000 |
| Distribution costs | $C_{distr} = k_{distr}*q_s$ | 28 |
| Overhead costs | $C_{OH} = k_{OH}*N_s*T/30$ | 20 000 |
| Employee costs | $C_{emp} = k_{emp}*N_s*EPS*T/30$ | 157 200 |
| Laundry costs | $C_{laundry} = 0$ | 0 |
| Repair costs | $C_{repair} = k_{repair}*t_s$ | 1600 |
| End-of-Life costs | $C_{EoL}=k_{EoL}*q_s*CR$ | 1781 |
| | **Profit ($\pi_s$)** | 319 391 |

Stipulating the same profit for the rental business model allows calculation of the required number of rental transactions ($t_r$) and consequent number of jackets produced ($q_r$) in the rental model. Considering that revenues minus costs should add up to the profit, $\pi_r$, the following equation can be set up:

*Equation 4:*

$$\pi_r = RE_r + RE_{r,2nd} - C_{prod} - C_{distr} - C_{OH} - C_{emp} - C_{laundry} - C_{repair} - C_{EoL}$$

Some costs and revenues depend directly on $t_r$, while the rest depend on the number of jackets produced ($q_r$), or the number of stores ($N_r$). However, each revenue and cost can be expressed in terms of the rental transactions, $t_r$, by expressing the relations between $t_r$, $q_r$ and $N_r$ by means of a coupling factor ($f$) for each cost or revenue, which have been derived in Appendix B of Paper IV and are summarised in Table 7. The coupling factors enable us to rewrite Equation 4 as the following:

*Equation 5:*

$$\pi_r = (f_1 + f_2 - f_3 - f_4 - f_5 - f_6 - f_7 - f_8 - f_9) * t_r$$

Solving Equation 5 for the rental transactions gives the number of transactions required to reach the profit defined as the functional unit.

The corresponding number of new jackets produced, $q_r$, needed to replace those sold 2nd hand can be derived via the following relation between $t_r$ and $q_r$ (detailed in Appendix B of Paper IV):

*Equation 6:*

$$q_r = \frac{U_r}{RL} * t_r$$

*Table 7: Revenues and costs in the rental model (according to the cost structure presented in section 3.4), connected to the number of transactions ($t_s$) by using coupling factors derived in Appendix B in Paper IV.*

| **Revenue or cost category** | **Revenue or cost expressed in terms of $t_r$** | **Coupling factor** |
|---|---|---|
| Revenues from sales transactions | $RE_r = f_1 * t_r$ | $f_1 = P_r$ |
| Revenues from 2nd hand sales | $RE_{r,\ 2nd} = f_2 * t_r$ | $f_2 = P_{2nd} * U_r/RL$ |
| Production costs | $C_{prod} = f_3 * t_r$ | $f_3 = k_{prod} * U_r/RL$ |
| Distribution costs | $C_{distr} = f_4 * t_r$ | $f_4 = k_{distr} * U_r/RL$ |
| Overhead costs | $C_{OH} = f_5 * t_r$ | $f_5 = k_{OH} * (T/30) * U_r/(E_r * T * SS)$ |
| Employee costs | $C_{emp} = f_6 * t_r$ | $f_6 = k_{emp} * (T/30) * EPS * U_r/(E_r * T * SS)$ |
| Laundry costs | $C_{laundry} = f_7 * t_r$ | $f_7 = k_{laundry}$ |
| Repair costs | $C_{repair} = f_8 * t_r$ | $f_8 = k_{repair}$ |
| End-of-life costs | $C_{EoL} = f_9 * t_r$ | $f_9 = k_{EoL} * CR * U_r/RL$ |

The results of the coupling phase are summarised in Table 8. The number of transactions and amount of production for each business model are the parameters fed into the subsequent phase, the life cycle inventory.

*Table 8: Basis of comparison and number of customer transactions and jackets produced in each business model*

|  | **Sales** | **Rental** |
|---|---|---|
| **Profit ($\pi$)** | 319 391 SEK | 319 391 SEK |
| **Transactions ($t$)** | 200 | 1108 |
| **Jackets produced ($q$)** | 200 | 28 |

### 6.2.3. Life cycle inventory and impact assessment

The number of transactions and required amount of production in each business model were used to build the LCI, perform the LCIA and interpret the results. Conventional LCA methodology was applied and OpenLCA software was used.

For the LCI, data were collected as described in in Paper IV and a life cycle model was built by considering all environmentally relevant flows, scaled according to the defined functional unit. Detailed LCI and the related data sources and modelling choices are presented in Appendix A of Paper IV.

Weighting was employed to identify the impact categories that contributed the most to overall impacts (detailed LCIA results are presented in Paper IV). Applying the ReCiPe (H,A) and Ecological Scarcity endpoint methods showed that the dominant impact category was climate change (for details, see Appendix C in Paper IV). The impact scores for climate change are shown in Figure 11 for each business model, divided into different life cycle stages. As seen, comparing the sales and rental models, impacts were shifted from production to use phase. Particularly, potential impact from energy intense production processes like 'spinning & weaving' and 'dyeing & drying' were reduced in the rental model since fewer new jackets were needed. In contrast, the rental model gave an eight-fold increase in potential impacts related to the use phase, mostly due to increased customer transport. Overall, however, the total score for climate change was 43% lower in the rental model, meaning that it represents a more decoupled business.

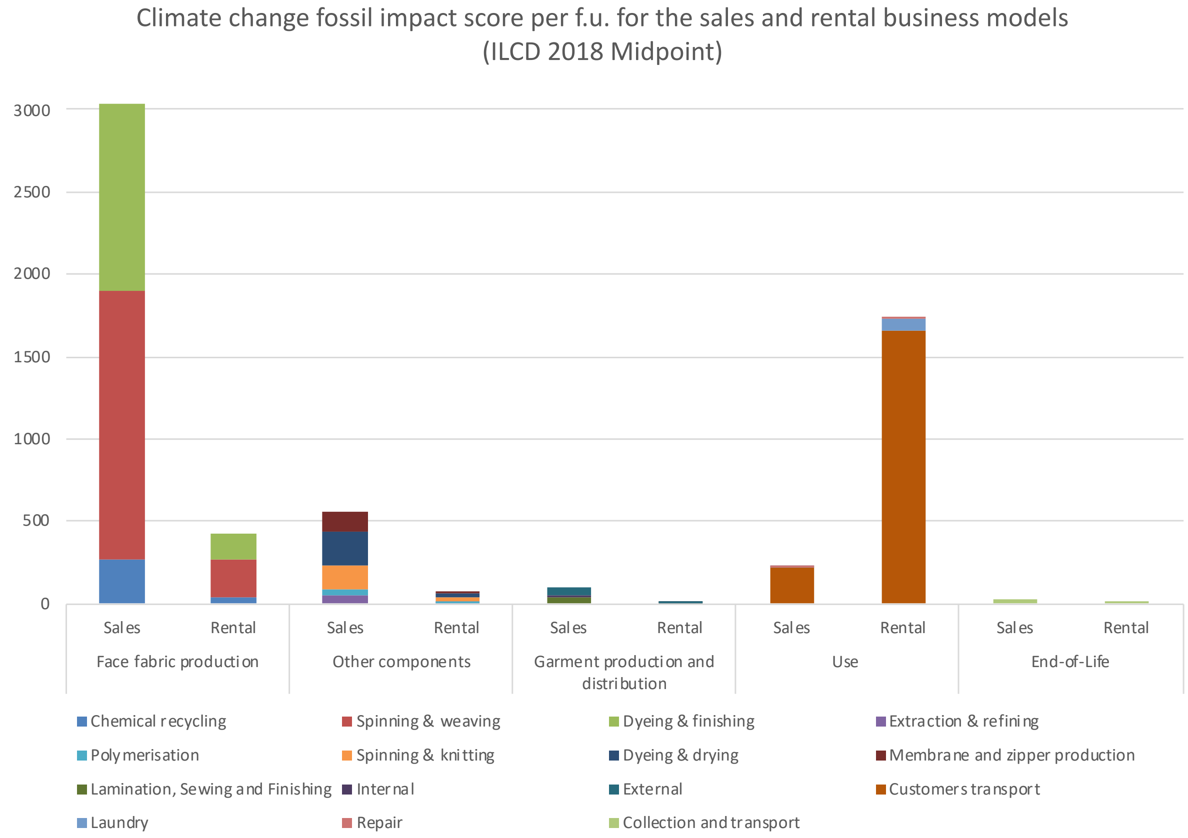

*Figure 11: Impact score per functional unit for climate change, divided into aggregated life cycle phases.*

### 6.2.4. Sensitivity analysis

A sensitivity analysis was performed to investigate the effects of changing selected business parameters, uncertain parameters and parameters for dominating life-cycle phases. The sensitivity analysis was done using weighted results from applying the ReCiPE (H,A) Endpoint method. The results for the baseline scenario, presented in the previous section, is shown on top of Figure 12, while the sensitivity to selected parameters are shown beneath.

As shown by Figure 12, results are highly sensitive to the rental price. With a 50% lower rental price, more rental transactions are needed to generate the same profit of the sales business model, reversing the ranking order between the business models. Conversely, a 50% higher rental price makes the rental business model even more preferable than the sales business model, as compared to the base case.

As also evident from Figure 12, a hybrid business model, where customers are offered to buy the rented jacket at a reduced price, is less environmentally preferable than a sales business model. On the other hand, keeping the jackets in the rental business for longer and renting them until the they reach the end of their technical lifetime reduces the environmental impact of the rental business model, albeit not significantly. In addition, the rental efficiency (i.e., the average share of jackets in the rental business being rented at a given time) significantly influences results and, if too low, may even reverse the ranking between the business models.

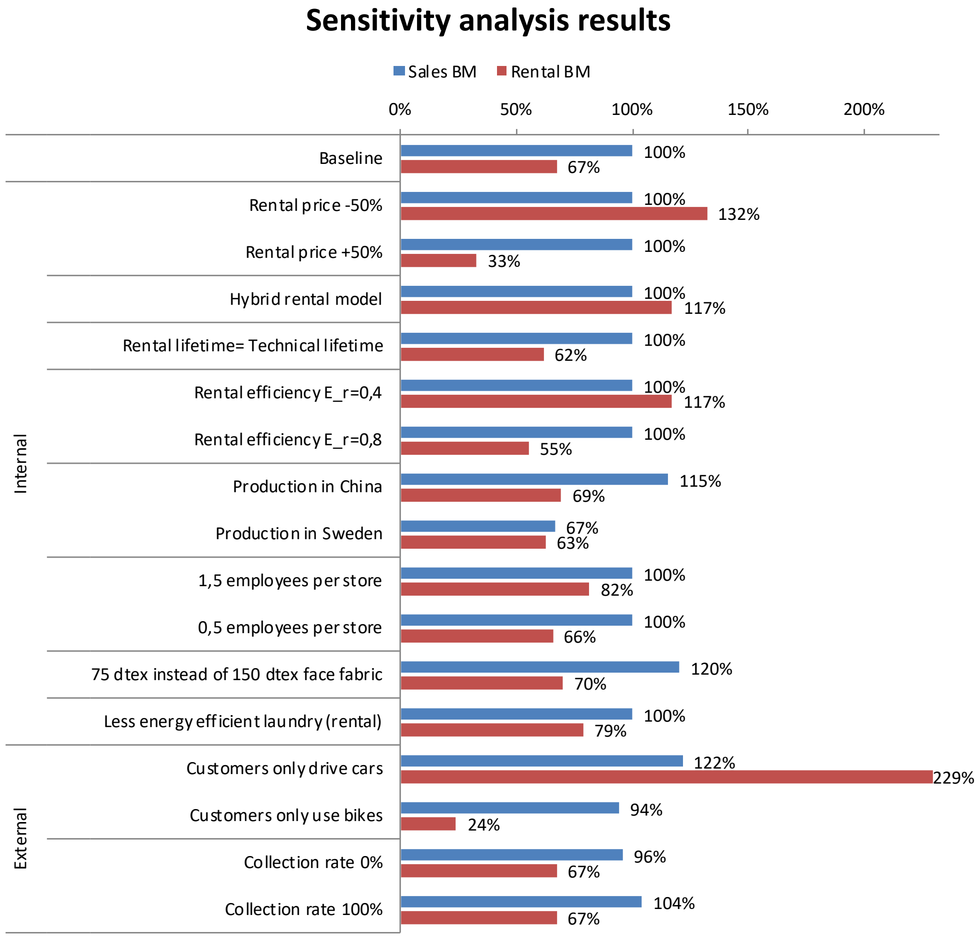

*Figure 12: Sensitivity analysis with respect to selected parameters, shown as single score from results from ReCiPE (H,A) Endpoint and are normalized the sales business model baseline scenario. Tested parameters are shown on the vertical axis, divided into internal and external factors, reflecting parameters which can be directly managed or only influenced by the company.*

Supplier choice can make a large difference, particularly in the sales model. If textile production is moved from Japan to Sweden, with a lower share of fossil fuels in the electricity mix, production impacts are reduced. If, conversely, textile production uses a high fossil electricity mix, here exemplified by production in China, the environmental impacts from production are increased, which is turn has a negative effect on both business models, but the sales model to a larger extent.

Another business-related aspect is the number of employees per store. If increased from 1 to 1.5 employees per store, this has a moderately negative effect on the results for the rental model. Conversely, a decrease only slightly reduces impacts.

In addition to business model choices, the company can alter the product design, e.g., by altering the quality of the textile. A fabric with a higher fibre density (75 dtex instead of 150 dtex) increases the environmental scores for the sales model, since more energy is required to achieve the higher density fabric. The rental model is only slightly affected because of the lower production volume. Furthermore, the use of less energy efficient laundry (washing twice as often, at 60-degree temperature, with electricity with a high share of fossil fuels) in the rental business model has a moderately negative effect on the rental business model.

In summary, several internal aspects that the company can directly control significantly affect the results. Some of these internal aspects were related to the business model setup, such as rental price or supplier choice, while others related to product design and maintenance.

In addition to factors that the company can directly control, there are relevant external aspects that can only be indirectly influenced. Figure 5 shows that the results were highly sensitive to customer's mode of transportation, much more in the rental model, which involves twice as much transportation by customers as the sales model. When all customers drive cars (instead of the 20% in the baseline model), the impact scores for the rental model more than double, reversing the ranking between the business models. Contrarily, when all customers use bikes (instead of the 20% in the baseline model), the environmental performance of the rental model is significantly superior to the sales model. In addition, the rate at which jackets are collected for recycling at end-of-life is under direct control of customers and can only be influenced by the company. As shown by Figure 5, the outcome of both the rental and the sales business models are influenced by collection rate, but to a limited degree

### 6.2.5. Recommendations to the case company

Answering the specific questions posed in section 3.1, the BM-LCA results show that the rental business model can lead to an overall better environmental performance compared to a sales business model, while maintaining the company's profit level.

The main hotspots in the sales business model include the production phase, due to energy intensive processes related to the large production volume, particularly regarding the face fabric. In the rental business model, environmental impact is instead dominated by the use phase, mainly caused by the increase in customers' transport to pick up and return jackets.

The sensitivity analysis showed that the rental model does not unambiguously result in a lower overall impact score, since some parameters strongly affect the environmental performance of the rental business model. While some of these are outside the company's control, they can still be managed. An example includes efforts to influence customers transportation habits towards sustainable transport modes. For the same reason, store location is an important factor within the company’s control which could for example be located in proximity to public transportation.

Other business factors within company control include the option to offer hybrid forms of rental services (selling rented jackets), which should be avoided since it represents the loss of potential revenues from repeated jacket rentals. The company should also set the rental price as high as possible, finding a balance between market considerations (e.g., demand and customers’ willingness to pay) and sustainability ambitions. Similarly, the rental efficiency should be maximised.

# Chapter 7 Discussion

The purpose of this dissertation was to investigate how guidelines for resource efficiency can be better grounded in empirical evidence. This was achieved by using of quantitative environmental assessments. Two research questions were addressed. The first (RQ1) concerned the formulation of empirically grounded guidelines for resource efficiency. The second (RQ2) concerned the environmental consequences of following guidelines and heuristics in two particular cases, namely 3D printing in the automotive industry and alternative business models (specifically renting of jackets). What follows will discussed how each respective research question addressed the research gaps, and compare the research contributions to literature.

## 7.1. Empirically grounded guidelines for resource efficiency

The synthesis of assessment studies in Paper I, summarised in Chapter 4, led to the formulation of empirically grounded, product-based guidelines for resource efficiency, presented in Figure 6. Table 9 compares the product-based guidelines to others in literature, by ordering guidelines according to their conceptual or empirical basis (on the vertical axis) and whether they rank their proposed measures or not (on the horizontal axis). Guidelines within each of the four resulting categories are further divided according to their intended application, namely 1) general guidelines, 2) guidelines for design applications, 3) guidelines for business or 4) guidelines specific to other applications like policy or manufacturing. The table shows that what sets the product-based guidelines apart from most other guidelines is the empirical and bottom-up basis, and that there is no ranking of the proposed measures. Further, the guidelines are general in the sense that they are applicable to all kinds of products (in all sectors and with many different characteristics) and are valid for many, if not all, types of applications. There are several pros and cons with such features compared to other guidelines, as will be discussed.

Many guidelines for resource efficiency are formulated from a top-down and conceptual perspective, meaning that they tend not to be built on case-based quantitative environmental assessments. The measures suggested by conceptually based guidelines tend to rest on idealised descriptions of reality, providing no clear understanding of the potential environmental consequences of applying each measure. Conceptually based guidelines may be intuitive and offer a clear overview of the suggested measures. Nevertheless, in order to be meaningful to support decision-making, each measure must be translated for every new case, to reflect the more complex reality. Furthermore, because conceptually based guidelines are built on a simplified image of industry and the situations where they aim to guide decisions, their validity and limitations in different contexts is generally unclear. It can, for example, be unclear whether the suggested measures are valid only for a certain type of product or a certain type of actor. The product-based guidelines avoid such problems because they are based on empirical environmental assessments and take into account the different types of products for which each suggested measure is suitable. Synthesis and generalisation are time-consuming and challenging, as is the formulation of these guidelines in a user-friendly manner that avoids being too complex or intricate. Nevertheless, a bottom-up approach can better reflect reality and enable the formulation of guidelines that are empirically grounded and more appropriate for guiding decision-making.

*Table 9: Overview of different guidelines for resource efficiency, grouped according to whether they have a theoretical or empirical basis and whether the suggested measures are generally ranked or not.*

| | **Ranking** | | **No ranking** | |
|---|---|---|---|---|
| **Conceptual basis** | **General guidelines**<br>- ReSOLVE (EMF, 2015a)<br>- Waste hierarchy (EC, 2008)<br>- 10R (Reike et al., 2018) | **Design guidelines**<br>- ReX taxonomy (Sihvonen and Ritola, 2015)<br>- 3R (Gehin et al., 2008)<br>- 6R (Yan and Feng, 2013) | **General guidelines**<br>n.a. | **Design guidelines:**<br>-Circular design guidelines (Bovea and Perez-Belis, 2018)<br>- Design for CBMs (Moreno et al., 2016)<br>- Life Cycle Design (Vezzoli and Manzini, 2008) |
| | **Business guidelines**<br>- CBM typology (Pieroni et al., 2020)<br>- PSS (Tukker, 2015) | **Policy/manufacturing guidelines**<br>- 9R (Potting et al., 2017) | **Business guidelines**<br>- CBM patterns (Lüdeke-Freund et al., 2018)<br>- CBM strategies (Bocken et al., 2016) | **Policy/manufacturing guidelines**<br>- ResCoM (Rashid et al., 2013)<br>- Circular Strategies Scanner (Blomsma et al., 2019) |
| **Empirical basis** | **General guidelines**<br>n.a. | **Design guidelines**<br>n.a. | **General guidelines**<br>Product-based guidelines (Paper I) | **Design guidelines:**<br>- Tool for resource efficient products (Willskytt and Brambila-Macias, 2020) |
| | **Business guidelines**<br>n.a. | **Policy/manufacturing guidelines**<br>n.a. | **Business guidelines**<br>n.a. | **Policy/manufacturing guidelines**<br>- Green principles for... stationary energy storage (Arbabzadeh et al., 2016)<br>- ...mobile energy storage (Arbabzadeh et al., 2019)<br>- ...lightweighting vehicles (Lewis et al., 2019) |

Another distinguishing trait of the product-based guidelines is that recommendations are formulated based on product characteristics rather than an overall ranking of measures. Kirchherr et al. (2017) argues that ranking of measures is crucial in promoting radical rather than incremental change and to avoid greenwashing by companies that, for example, collect and send their products for recycling, claiming that this makes them fully circular or resource efficient. Nevertheless, while such general rankings might in many cases accurately predict which measure would give the most environmental benefit, they do not apply generally as there are exceptions that can be identified by empirical assessments. For instance, the EU waste hierarchy can be overturned in specific cases, and within the directive there is a call for detailed assessments that are necessary for more factual decision support (EC, 2008; Manfredi et al., 2011). Additionally, it is sometimes not meaningful to rank measures since they can be interdependent (Blomsma and Brennan, 2017). An example can be found in the study of André et al. (2019) who show that reuse of laptops also leads to increased recycling. Several of the guidelines in Table 9, like the R-hierarchies or the EU waste hierarchy, rank measures more or less explicitly. In comparison, the product-based guidelines lack ranking due to the empirical basis that enabled taking into account details of product characteristics and the potential trade-offs associated with each proposed measure.

Other guidelines that do not rank measures include ResCoM, which recommends products to be designed for all possible resource efficiency measures simultaneously. Another example is the circular strategies scanner which acknowledges that measures can be implemented in ‘circular configurations’ (i.e., combinations of measures working is sequence or parallel) and that there can be trade-offs and synergies between measures. A parallel can be drawn between the guidelines for design of CBMs (Moreno et al., 2016) and the empirically grounded product-based guidelines. While the latter uses a synthesis of quantitative assessments to formulate which measures are suitable for what type of products, Moreno et al. (2016) synthesise qualitative literature to formulate circular design guidelines based on different business considerations.

While the business guidelines by Bocken et al. (2016) and Lüdeke-Freund et al. (2018) do not rank their proposed measures, their recommendations are conceptually based and thus lack an empirical grounding. By contrast, some guidelines are based on environmental assessments, such as the green principles for stationary and mobile energy storage (Arbabzadeh et al., 2016; Arbabzadeh et al., 2019) and for vehicle lightweighting (Lewis et al., 2019). However, these guidelines are specific to one application, unlike the product-based guidelines whose recommendations indicate what measures are suitable for what type of product. This implies that the suggested measures can be applied to products in widely different sectors, as long as the products share similar characteristics such as being consumable or active and durable. For example, a car and a house share the trait of having an energy-intensive use phase, meaning that reducing use of auxiliary materials and energy is a suitable measure for both types of products. Consequently, the product-based guidelines are general while still reflecting reality where the suitability of measures depends on product characteristics and context.

The only other guidelines in Table 9 that take into account product characteristics are specific to design. An example is the life cycle design guidelines of Vezzoli and Manzini (2008), which, while built on a conceptual basis, give different recommendations for different product types. A further example is the circular design guidelines by Bovea and Perez-Belis (2018), who use product characteristics to determine which specific guidelines to employ in order to achieve a circular design. The tool for resource efficient eco-design by Willskytt and Brambila-Macias

(2020) was also formulated in terms of product characteristics, since it directly expands on the product-based guidelines, specifically for design purposes.

While the empirically based guidelines bring several advantages, achieving this empirical basis is associated with some challenges. Synthesising the assessment studies was difficult because of the diversity of the collected studies, in their approach, stated goals, system boundaries and presentation of results. The analysis and comparison of studies was aided by representing assessment results in a semi-quantitative manner (see section 3.3). This allowed simplification while still retaining enough detail to be able to investigate which measures correlated with reduced environmental impacts for different types of products. However, this simplification also entailed a loss of nuance and introduced a degree of uncertainty in interpreting each case study's results. In addition, the empirical status of some of the synthesised assessment studies can be questioned. This is because several studies compare a conventional case with a hypothetical case formulated, for example, based on a measure from the R-hierarchies. Future syntheses should favour assessments built on real cases to strengthen the empirical basis of the guidelines.

In conclusion, it was shown that more well-founded guidelines could be based on synthesis of environmental assessments in order to recommend measures for key product types. While the guidelines were general, what set them apart from guidelines with an overall ranking of measures was that they are clear on what measures are suitable for each type of product and which trade-offs can occur.

## 7.2. Guiding the development and application of 3D printing

Existing heuristics regarding the potential environmental benefits of metal 3D printing in the automotive industry were scrutinised using a prospective LCA, presented in Paper II and summarised in Chapter 5. The results can guide decision-making for various actors, such as developers of metal 3D printing technology or manufacturers intending to use the technology in their processes. For instance, the study revealed that 3D printing offered potential future reductions of a truck's life cycle environmental impacts. Importantly, however, this is only true under the right conditions. First, green electricity must be used to reduce the negative effects of the highly energy-intensive printing process, which is relevant for any application of the technology by manufacturers. Second, the materials that can be used for printing today tend to be materials with high environmental impact like nickel alloys, titanium alloys or stainless steel. Hence, to realise environmental benefits, there must be developments towards printing with materials like low-alloy steel instead. Such a development is uncertain, however, since the experts operating the 3D printers or designing the parts to be printed tend to focus on optimising part structure and material properties rather than the reduction of environmental impacts. Nevertheless, with future diffusion and maturity of the technology, there may be incentives to use cheaper materials, which may in turn coincide with materials of lower environmental impact. This further supports the necessity of environmental assessments that highlight the ways in which the technology's environmental impacts can be reduced. This is particularly true for metal 3D printing, which is in an early stage of adoption. Guidance from relevant environmental assessments at this stage can make a large difference in a future with large scale adoption (Saade et al., 2020).

As a full prospective life cycle assessment of an application of 3D printing, the study in Paper II was one of the first of its kind, so there is a limited number of studies in literature to compare. Examples include a study by Mami et al. (2017) which is similar and concludes that 3D printing of airplane components provides life cycle environmental improvements due to lightweighting. Kamps et al. (2018) also confirm that lightweighting of components is a key factor for reducing energy consumption. Conversely, in applications without energy savings from lightweighting, 3D printing is less environmentally beneficial, e.g., for high-speed gears in wind turbines as shown by Liu et al. (2018), although the study does not consider technological developments. Conversely, a study by Huang et al. (2017) concludes that environmental improvements can be made in the case of injection moulds using mature 3D printing technology.

In summary, while there are significant expectations on the potential environmental improvements from 3D printing, prospective LCA has here been used and discussed to scrutinise such expectations. It was shown that metal 3D printing may reduce the life cycle impacts of trucks, but only under certain conditions (green electricity and low-impact materials).

## 7.3. Environmental assessments of business models

In order to test the potential of alternative business models like CBMs to decouple economic performance from environmental impact, a method dubbed BM-LCA was developed for the purpose (in Paper III) and is discussed in section 7.3.1. Furthermore, the method was applied for the environmental assessment of two business models of a Swedish apparel company (in Paper IV). The assessment served the dual purpose of assessing and comparing renting and selling of shell jackets, as well as testing and demonstrating the developed method and is discussed in section 7.3.2.

### 7.3.1. Methods for assessing business models

The key innovation by BM-LCA on the LCA method is the possibility to quantify the environmental consequences of different ways of making money. This is achieved by taking the business model itself as the object of analysis and using economic performance as the basis of comparison, expressed in a profit-based functional unit. By contrast, mainstream LCAs study products (or services), which means that their basis of comparison is the function or use of a physical product (or a product related to a service). Mainstream LCA applied to business models thus cannot reveal the direct environmental consequences of different ways of making money. Instead, such studies must estimate or assume how the business model will affect physical aspects of the product life cycle. Only through such estimates and assumptions can the environmental impacts eventually be calculated, as illustrated in Figure 13. Examples include a study by Diener et al. (2015). They used several physical traits to assess a potential shift to PSS for trucks, including a smaller truck fleet through better adaptation of vehicle size to specific transports, and design changes to the truck to enhance durability and changed maintenance and remanufacturing schemes. Similarly, a study by Bech et al. (2019) of T-shirts used design for improved durability to estimate environmental benefits of a new business model. Martin et al. (2021) estimated how frequently tools are used, in order to assess business models. All such assessments rely on estimates or assumptions of physical factors such as design, product lifetime, maintenance schemes and user behaviour.

In contrast, BM-LCA takes the business model itself as the object of analysis, why the basis of comparison is formulated as economic performance. This, in turn, allows pinpointing the direct consequences that different business decisions will have on environmental impact, as illustrated in Figure 13. For instance, altering an economic factor like product price has a direct effect on the number of customer transactions required in a rental business model and thus on environmental impacts. Consequently, a translation from business factors to environmental consequences via the physical aspects of the life cycle is avoided.[6] Results are thus less dependent on assumptions about, for example, how the lifetime of a product will change with the new business model or how often it will be used. As such, BM-LCA represents an important tool for investigating and understanding the environmental effects of different business models without confusing the analysis with other effects and without having to make assumptions about the use or durability of products.

In practice, the economic basis of comparison of BM-LCA was implemented as a profit-based functional unit. At a glance, this is radically different from the conventional functional unit based on product function or use. Nevertheless, in principle there are several similarities between the two. A product-based functional unit is a rough approximation of the actual function of the products (Baumann and Tillman, 2004). Two different products or services (even if they are versions of the same basic product/service) are as a rule not directly comparable because there are differences in quality, in how well they fulfil their function, or in whether one provides additional functions (such as superior aesthetics). Hence, when

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[6] Note that changes in the business model can affect the physical product system (see Figure 13). For example, a change in supplier can entail different production costs, which can be incorporated in the model equations. Conversely, for example, changing to a fast fashion model entails a different product design with different environmental impacts due to production.

defining a quantitative functional unit, most of these differences must be disregarded and the functional unit will thus be an approximation that reflects only the main function of the products/services. Some differences, like changes in product lifetime, may be captured by a product-based functional unit, but others, like a more or less beautiful product, are not measurable and cannot be captured. Similarly, the profit-based functional unit disregards some aspects of the business models being compared, such as added value in terms of, say, improved employee motivation or increased strategic fit (Geissdoerfer et al., 2018b). Instead, the function of a business model is boiled down to the main function of the business, which is the generation of profit (see section 6.1).

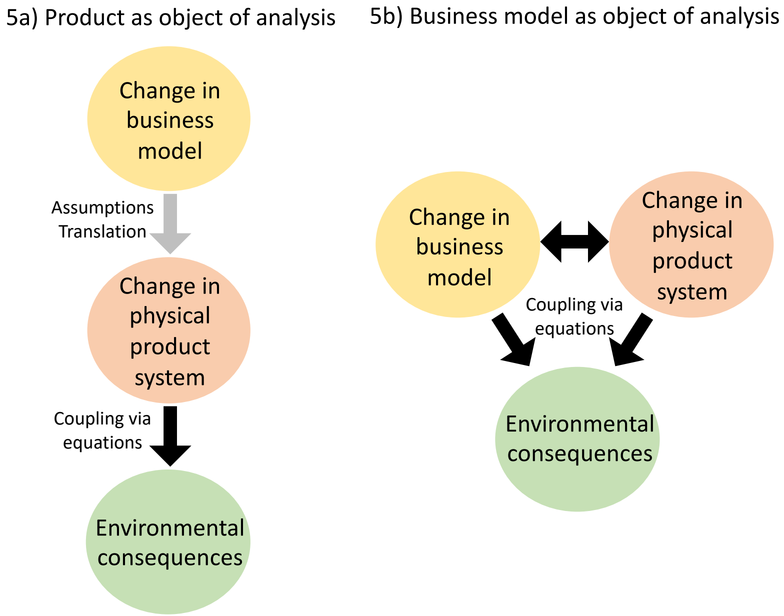

*Figure 13: Schematic of how business model changes are connected to their environmental consequences, depending on a product- or business model object of analysis. Either a) the environmental consequences of business must be translated via assumptions (grey arrow) on how business affects the physical product system, which in turn directly affects the environmental impacts (black arrow); or b) by coupling monetary and physical flows in the model equations, BM-LCA provides a way to directly see the effects of business on the environmental impacts.*

Comparing with previous assessments of business models, Table 10 presents a selection of studies, divided according to their object of analysis, whether they centre on products and product function or on business models and their economic performance. The table also separates studies according to the use of economic data in their modelling. Compared to environmental LCAs that do not take into account economic data, BM-LCA has a business focus and a profit-based functional unit which allows for a model that actually reflects the environmental consequences of different ways of making money. Additionally, the coupling of physical and monetary flows ensures that BM-LCA can be used to perform sensitivity analysis on business parameters by quantifying the environmental consequences of making different business choices. This is relevant for guiding business decisions towards decoupling economic performance from environmental impact. The main difference compared to studies that do use economic data in their models, like parallel LCA/LCC studies and eco-efficiency calculations, has already been discussed above, namely that such studies use products as their object of analysis and product function as their basis of comparison. Consequently, BM-LCA is unique in bringing business aspects into LCA to enable the environmental assessment of business models.

*Table 10: Examples of quantified environmental assessments of business models from literature, grouped according to their object of analysis and whether economic data is used in modelling or not.*

| | **Object of analysis (basis of comparison)** | | |
|---|---|---|---|
| | **Product (function of product)** | | **Business model (economic performance)** |
| **No economic data in modelling** | **Environmental LCA:**<br>- Bech et al. (2019)<br>- Chun and Lee (2017)<br>- Kerdlap et al. (2021)<br>- Martin et al. (2021)<br>- Zamani et al. (2017) | **LCA with qualitative economic perspective:**<br>- Barbieri and Santos (2020)<br>- Hoffmann et al. (2020)<br>- Tschiggerl et al. (2018) | n.a. |
| **Economic data in modelling** | **Parallel LCA and LCC:**<br>- Kaddoura et al. (2019)<br>- Zhang et al. (2018)<br>- Bi et al. (2015)<br>- Bi et al. (2017) | **Simulation-based tools:**<br>- Asif et al. (2016)<br>**Eco-efficiency:**<br>- Mendoza et al. (2019) | **BM-LCA** |

In essence, the key innovation of BM-LCA over conventional LCA is to shift the object of analysis from products to business models per se. The use of a profit-based functional unit enabled the comparison of business models from a company perspective, to identify the ways of making money that result in lower environmental burden.

### 7.3.2. Potential decoupling from alternative business models

In order to evaluate the potential of a rental business model to reduce environmental impacts while maintaining profitability, BM-LCA was applied on a case company providing jackets (in Paper IV, summarised in section 6.2). Results show that renting represents a more decoupled business model than the sales model. The main reason was that a jacket can be rented multiple times and generate more revenue in total compared to selling a jacket, thus reducing new production. However, results were sensitive to some key parameters that were identified via sensitivity analysis.

The application of BM-LCA enabled sensitivity analysis with respect to business parameters to investigate their influence on environmental impacts. For example, the study showed that an increase in the rental price reduces environmental impacts per functional unit, because fewer jackets and transactions are then needed to reach the same profit. A company implementing a rental model should thus set the rental price as high as possible without exceeding customer willingness to pay. Rental efficiency was another business-related parameter important for the environmental results, as a low efficiency represents an inefficient utilisation of the stock of rental jackets. To compensate, more rental transactions and jackets are necessary, thus reversing the ranking between the business models. Consequently, a company implementing a rental business model should aim to maximise rental efficiency in order to achieve decoupling.

Further, in the sales model, the dominating life cycle phase was the production of the jacket's face fabric. On the other hand, in the rental model the production impacts per functional unit were reduced greatly due to the lower number of jackets produced. The dominating phase in the rental model was instead customer transportation, why a potential way to reduce impacts is to influence customers to use more sustainable means of transportation. Other ways of decreasing impacts in the rental model are by increasing the jacket's technical lifetime via product design. Similarly, increasing the rental lifetime reduces the replacement rate of jackets which in turn reduces the need for producing new jackets. However, the benefits of prolonged life are significantly smaller than the overall environmental improvement from switching from a sales to a rental model or decreasing customer transportation impacts.

These considerations show that, because BM-LCA allows sensitivity analysis with regard to business parameters, it is a useful method for measuring the cause-effect relations between the economic and environmental elements of the business models in a real case. Thus, it is suitable for supporting the company's decision making, not only regarding new product design or production system (as a more mainstream LCA would) but mainly regarding how to adapt their business model for decoupling, for example by aiming for a higher rental efficiency.

To summarise, the application of BM-LCA was useful in shifting the perspective of the assessment to the case company and their business models. This showed that renting instead of selling of jackets has a lower environmental impact, while maintaining the level of profits. BM-LCA made possible the sensitivity analysis of business parameters, thus identifying, e.g., rental price and rental efficiency as key parameters determining environmental performance.

## 7.4. Actor perspective and LCA modelling

Resource efficiency guidelines are meant to be used by someone, be they product designers, business developers or policy-makers. Similarly, the results of environmental assessments are intended to be used, for instance, to support a decision or inspire a new solution. Consequently, when constructing the model and communicating the results of an assessment, it is crucial to have the intended audience in mind and consider what that audience can influence. For example, a designer can use LCA results to redesign a product for reuse or recycling, while a business developer should be able to use results for business model innovation. However, designers have influence over certain decisions but not others. For example, they may be able to directly influence material choices but may not have authority over reverse logistics. Likewise, a company can make changes within their own organisation, while they can only have more or less indirect influence over other actors like customers or suppliers. For instance, the assessment in Paper IV showed the environmental importance of business-related factors over which the company has direct control. Additionally, the study showed the importance of customer transport choices, over which the company has no authority but may still be able to influence in a desired direction. Such an analysis was enabled by the underlying concept of PCO, where transactions between different actors could be mapped and their interactions tracked.

It follows that models need to be constructed, and results presented, to be relevant to the actors that can most use the results. The same is true also when formulating guidelines, although many of those discussed in section 7.1 provide general guidelines with little consideration for how the recommended measures will be employed in practice. In this dissertation, examples of actor-adaptation include Paper II, where key take-aways were formulated for developers of 3D printing technology, for the company under study and for other manufacturers who may use 3D printing. The motivation behind the BM-LCA method presented in Paper III and the assessment study in Paper IV was the need for empirically-based guidance of business decisions towards decoupling. To that end, the model was built to generate results of relevance for a business audience. Furthermore, the choices that the company could make to improve decoupling were emphasised by carrying out sensitivity analyses for various business parameters. A similar approach was adopted by Löfgren (2012) who presents a way to adapt LCA studies to produce results relevant to manufacturing managers. Specifically, he adapts the system boundaries of the LCA model to reflect the life cycle implications of those actions, which can be directly controlled by manufacturing managers.

A further aspect of adaptation of LCA for different actors lies in the level of detail of the results presented. LCA results often have to be communicated to audiences with little or no experience of the methodology, such as policy-makers, business managers or consumers. It is thus crucial to convey key findings in a clear, relevant and concise manner. Both Paper II and IV employed the use of weighting methods to filter the assessment results in order to present and analyse the most relevant impact categories in further detail, thus avoiding to drown out key findings in less relevant results. While a drawback of using weighting is that it necessarily entails subjective value judgements, applying several methods that build on different weighting principles reduces the dependence on value judgements. This approach takes advantage of the benefits of weighting, while avoiding several of the drawbacks.

Finally, a note on the use of life cycle studies for supporting business activities. Piekarski et al. (2013) suggest that LCA can be used as an ‘entrepreneurial tool for business management and green innovations’. Particularly, LCA can support strategic planning (e.g., via supply-chain management), research and product development of sustainable products and production processes as well as the company’s marketing and social and environmental responsibility work. However, it can be argued that, because conventional LCA excludes business considerations, the environmental considerations will tend to be subordinated economic interests in any given decision situation (Nilsson-Lindén et al., 2019). In contrast, from the discussion on BM-LCA, in section 7.3, can be inferred that the developed method represents a new type of analysis that provides a more substantiated input to business decisions, by including relevant business considerations into the modelling. Furthermore, BM-LCA has uses beyond the management areas mentioned by Piekarski et al. (2013), namely in guiding the business model innovation process itself. For example, sensitivity analysis of business parameters can underpin what Sarasini and Linder (2018) call an organisation's 'reflexive activities' for monitoring, assessing and evaluating their practices in relation to sustainability transitions. These reflexive activities, and in turn BM-LCA, are key for achieving business model innovation towards environmentally sustainable business and warrant further research.

In short, an important characteristic of guidance of any kind is that it is adapted to the actor that is meant to be guided. Various aspects of the research in this dissertation were purposely adapted to be useful to actors, such as BM-LCA taking a company perspective. Another example is the product-based guidelines that, while generally applicable, are formulated to be clear on what measures are suitable for any given product. Overall, there is scope for further discussion and research on how to take into account different types of actors in environmental studies as well as on how the modelling and results of studies can be adapted to different actors.

## 7.5. Validity and reliability

Quantitative empirical case studies are the foundation of all the research presented in this dissertation, as described in 0. Paper II was an empirical case, Paper III presented a method for quantitative assessments, Paper IV applied the method in a real case, while Paper I synthesised quantitative assessments from literature. The validity of the research is thus strongly connected to this pervading empirical foundation. Particularly in the case of the empirically grounded guidelines, the underlying case studies enabled detailed analysis of resource efficiency measures. Synthesis of many studies then allowed the formulation of the guidelines presented in Chapter 4. These are generalised to any resource efficiency measure and a number of key product characteristics. However, they still take details into account by considering the identified trade-offs so that one measure is not always environmentally preferable over another, but depends rather on the case in question. The guidelines thus have a stronger foundation in empirical reality compared to more common theoretically-based guidelines.

There are limitations to the general validity of single assessment studies (Baumann et al., 2002). The focus on a single case can enable the study of the details relevant to that specific context at the cost of generalisability. Generalisation can still be achieved by using an in-depth analysis and understanding enabled by an empirical case study (Flyvbjerg, 2006). A common way of achieving such understanding in LCA is through sensitivity analysis, which allows the investigation of how variations in model parameters and processes affect the results (Baumann and Tillman, 2004). In addition, sensitivity analysis can be used to test the reliability and robustness of the results, which can also be enhanced, e.g., by making conservative assumptions that favour the reference option rather than the alternative (employed in Papers II and IV). Regarding the use of case studies to formulate guidelines, general guidelines tend to be formulated on a conceptual basis and can often be overturned in specific cases. In contrast, the product-based guidelines are valid for any type of application, depending instead on the characteristics of the product in question. This was made possible by synthesising a large number of individual case studies, although it should be noted that the validity of such guidelines depends on the validity of the synthesised assessment studies.

# Chapter 8 Conclusions

This dissertation has challenged prevalent guidelines for resource efficiency that are conceptually based, and which often rank their suggested measures. Empirically based guidelines were formulated by synthesising environmental assessments and indicating for which product types that different measures are effective and where there can be trade-offs. Furthermore, this dissertation used environmental assessments to test expectations placed on 3D printing and alternative business models to reduce environmental impacts. By taking into account detailed and complicated flows and connections within industry, it was possible to bring nuance to such expectations, e.g., of the benefits of 3D printing in the automotive industry. Additionally, a new method, was developed that takes business models as the object of analysis and uses their economic performance as the basis of comparison. Business Model LCA was used to compare the environmental consequences of different business models in the case of comparing a rental and a sales business model for a Swedish apparel company.

In more detail, it was shown that empirically based resource efficiency guidelines can be formulated in terms of key product characteristics rather than a general ranking of measures. An example of a recommendation is that reuse (or sharing and other ways of using more of a product's technical lifetime) is only suitable for durable products that are typically discarded before being worn out. Of the trade-offs that were found to potentially affect the outcome of measures, an example is that, for active durable products where there is a technological development towards use-phase efficiency, there is a trade-off between use-phase efficiency and benefits from use extension. In summary, the following product characteristics were found to be key in determining what measures are suitable and when they are effective. First is whether products are consumable, divided into disposables and products used dissipatively. Second is whether products are durable, which can be active or passive products. Durable products can further be distinguished by whether they are typically used for their full technical *lifetime* or *used infrequently* and *typically discarded before being worn out.* The final key characteristics for durables are the *pace of development for a product* (relevant for durable, active products) and whether *functionality remains* after the end of use of a product. An underlying cause to the importance of several key product characteristics is what *life cycle phase dominates* environmental impact and resource use (e.g., often the use phase for active products). In addition, the literature showed that *product complexity* is of key importance.

Regarding the potential environmental improvements from realising heuristics, one of the studies concerned metal 3D printing. A prospective LCA study was carried out to investigate the potential for environmental improvements from redesigning a light truck engine for 3D printing, compared to conventional manufacturing. Results showed that, in its present state, applying 3D printing could lead to increased life cycle environmental impacts. Nevertheless, there is potential for environmental improvements in the future, but only under certain conditions. Green electricity must be used for the 3D printing process and there must be technological development to allow printing using materials with low environmental impact, such as low-alloy steel. The possibility of printing larger components would enable the technology to be used in more applications where the strengths of 3D printing could be leveraged. As such, a case based environmental assessment proved to provide nuance to the environmental expectations of 3D printing in the automotive industry and to identify an environmentally preferred direction for technology development and application.

The second area for scrutinising environmental expectations involved the potential of rental business models to decouple economic performance from environmental impact. BM-LCA was developed to take business models themselves as the object of analysis. Furthermore, the economic performance of a company's business models is taken as the basis of comparison. This enabled the coupling of monetary and physical flows which in turn allowed the investigation of the environmental consequences of different business decisions. The method is the first of its kind for comparing the environmental performance of business models.

BM-LCA was applied to a real case, in order to compare renting and selling of polyester jackets. The results showed that the rental model led to lower environmental impacts overall compared to the sales model, while still maintaining the same economic performance. The main reason for this was that the company can get more money out of each jacket by repeatedly renting it instead of selling it only once. However, the results were sensitive to the sustainability of the transport and energy systems, as well as to business model parameters like rental price and rental efficiency. A key take-away was the need for careful business planning for ensuring the decoupling potential of renting, where certain business practices, like letting customers purchase their jackets after renting them, had particularly negative implications for decoupling. Such sensitivity analysis of business parameters represents a new type of analysis not possible with conventional LCA. In summary, case based environmental assessment turned out to provide nuance and guidance to the promise of decoupling from rental business models, enabled by a new method developed for the purpose.

In conclusion, two key contributions and innovations of this dissertation can be emphasised. The first is the formulation of empirically grounded guidelines based on key product characteristics. Second is the formulation and testing of BM-LCA, a method for assessing the decoupling of business from environmental impact.

## 8.1. Future research

The scope for expanding and developing the research presented in this dissertation is presented in detail in Papers I–IV.

On an overall level, there is a further need for empirically based guidelines. Additional synthesis of high-quality assessments of resource efficiency solutions can be used in order to refine and better substantiate the product-based guidelines.

The synthesis approach can also be used to formulate more directed guidelines in specific contexts, such as for policy-making or business guidance. On the other hand, there is still a need for further assessments to evaluate the environmental efficacy of guidelines and heuristics, particularly in cases describing and revealing the full complexity of real industrial practice. This is true in regard to the investigation of resource efficiency solutions in general.

There is also a need for more environmental assessments of particular solutions like 3D printing, using updated data as the technology develops and is applied in other contexts in addition to the automotive industry.

BM-LCA can also be further developed according to the needs of businesses and applied to more cases, investigating various business models aiming to decouple economic performance from environmental impact.

# References

Allwood, J. M., Ashby, M. F., Gutowski, T. G., Worrell, E., 2011. Material efficiency: A white paper. *Resources, Conservation and Recycling, vol. 55*(3), pp. 362-381. doi: 10.1016/j.resconrec.2010.11.002

André, H., Ljunggren Söderman, M., Nordelöf, A., 2019. Resource and environmental impacts of using second-hand laptop computers: A case study of commercial reuse. *Waste Management, vol. 88*, pp. 268-279. doi: 10.1016/j.wasman.2019.03.050

Andrews, C. J., 2000. Building a Micro Foundation for Industrial Ecology. *Journal of Industrial Ecology, vol. 4*(3), pp. 33-51. doi: 10.1162/108819800300106375

Arbabzadeh, M., Johnson, J. X., Keoleian, G. A., Rasmussen, P. G., Thompson, L. T., 2016. Twelve Principles for Green Energy Storage in Grid Applications. *Environ Sci Technol, vol. 50*(2), pp. 1046-1055. doi: 10.1021/acs.est.5b03867

Arbabzadeh, M., Lewis, G. M., Keoleian, G. A., 2019. Green principles for responsible battery management in mobile applications. *Journal of Energy Storage, vol. 24*. doi: 10.1016/j.est.2019.100779

Arvidsson, R., Kushnir, D., Molander, S., Sandén, B. A., 2016. Energy and resource use assessment of graphene as a substitute for indium tin oxide in transparent electrodes. *Journal of Cleaner Production, vol. 132*, pp. 289-297. doi: 10.1016/j.jclepro.2015.04.076

Arvidsson, R., Tillman, A.-M., Sandén, B. A., Janssen, M., Nordelöf, A., Kushnir, D., Molander, S., 2017. Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. *Journal of Industrial Ecology, vol. 22*(6), pp. 1286-1294. doi: 10.1111/jiec.12690

Asif, F. M. A., Lieder, M., Rashid, A., 2016. Multi-method simulation based tool to evaluate economic and environmental performance of circular product systems. *Journal of Cleaner Production, vol. 139*, pp. 1261-1281. doi: 10.1016/j.jclepro.2016.08.122

ASTM International, 2012. Standard Terminology for Additive Manufacturing Technologies (Withdrawn 2015) ASTM F2792-12a. West Conshohocken, PA, USA: ASTM International.

Ayres, R. U., Ayres, L., 2002. *A handbook of industrial ecology*. Cheltenham, UK: Edward Elgar Publishing.

Bakker, C., Den Hollander, M., van Hinte, E., Zijlstra, Y., 2014. *Product that Last. Product Design for Circular Business Models*. Delft, Netherlands: TU Delft.

Barbieri, R., Santos, D. F. L., 2020. Sustainable business models and eco-innovation: A life cycle assessment. *Journal of Cleaner Production, vol. 266*. doi: 10.1016/j.jclepro.2020.121954

Baumann, H., 2008. *Simple material relations handled by complicated organisation by or 'How many (organisations) does it take to change a lightbulb?'*. What is an organization? Materiality, Agency and Discourse. May 21-22, 2008, HEC Montréal, Université de Montréal, Quebec, Canada.

Baumann, H., 2012. *Using the life cycle approach for structuring organizational studies of product chains*. 18th conference of the Greening of Industry Network. 21-24 October 2012.

Baumann, H., Boons, F., Bragd, A., 2002. Mapping the green product development field: engineering, policy and business perspectives. *Journal of Cleaner Production, vol. 10*, pp. 409-425. doi: 10.1016/S0959-6526(02)00015-X

Baumann, H., Tillman, A.-M., 2004. *The Hitchhiker's Guide to LCA: An orientation in Life Cycle Assessment Methodology and Application*. Lund, Sweden: Studentlitteratur.

Bech, N. M., Birkved, M., et al., 2019. Evaluating the Environmental Performance of a Product/Service-System Business Model for Merino Wool Next-to-Skin Garments: The Case of Armadillo Merino®. *Sustainability, vol. 11*(20). doi: 10.3390/su11205854

Bi, Z., De Kleine, R., Keoleian, G. A., 2017. Integrated Life Cycle Assessment and Life Cycle Cost Model for Comparing Plug-in versus Wireless Charging for an Electric Bus System. *Journal of Industrial Ecology, vol. 21*(2), pp. 344-355. doi: 10.1111/jiec.12419

Bi, Z., Song, L., De Kleine, R., Mi, C. C., Keoleian, G. A., 2015. Plug-in vs. wireless charging: Life cycle energy and greenhouse gas emissions for an electric bus system. *Applied Energy, vol. 146*, pp. 11-19. doi: 10.1016/j.apenergy.2015.02.031

Blomsma, F., 2018. Collective 'action recipes' in a circular economy – On waste and resource management frameworks and their role in collective change. *Journal of Cleaner Production, vol. 199*, pp. 969-982. doi: 10.1016/j.jclepro.2018.07.145

Blomsma, F., Brennan, G., 2017. The Emergence of Circular Economy: A New Framing Around Prolonging Resource Productivity. *Journal of Industrial Ecology, vol. 21*(3), pp. 603-614. doi: 10.1111/jiec.12603

Blomsma, F., Pieroni, M., et al., 2019. Developing a circular strategies framework for manufacturing companies to support circular economy-oriented innovation. *Journal of Cleaner Production, vol. 241*. doi: 10.1016/j.jclepro.2019.118271

Blüher, T., Riedelsheimer, T., Gogineni, S., Klemichen, A., Stark, R., 2020. Systematic Literature Review—Effects of PSS on Sustainability Based on Use Case Assessments. *Sustainability, vol. 12*(17). doi: 10.3390/su12176989

Bocken, N. M. P., de Pauw, I., Bakker, C., van der Grinten, B., 2016. Product design and business model strategies for a circular economy. *Journal of Industrial and Production Engineering, vol. 33*(5), pp. 308-320. doi: 10.1080/21681015.2016.1172124

Bocken, N. M. P., Olivetti, E. A., Cullen, J. M., Potting, J., Lifset, R., 2017. Taking the Circularity to the Next Level: A Special Issue on the Circular Economy. *Journal of Industrial Ecology, vol. 21*(3), pp. 476-482. doi: 10.1111/jiec.12606

Bocken, N. M. P., Short, S. W., Rana, P., Evans, S., 2014. A literature and practice review to develop sustainable business model archetypes. *Journal of Cleaner Production, vol. 65*, pp. 42-56. doi: 10.1016/j.jclepro.2013.11.039

Böckin, D., Goffetti, G., Baumann, H., Tillman, A.-M., Zobel, T., 2020. *Environmental assessment of two business models - a life cycle comparison between a sales and a rental business model in the apparel sector in Sweden* (2020:2). Gothenburg, Sweden: Chalmers University of Technology. https://research.chalmers.se/publication/519800

Boulding, K. E., 1966. The economics of the coming spaceship earth. In Environmental quality in a growing economy, ed. H. Jarrett, (pp. 3-14). Washington D.C., United States: Resources for the Future.

Bovea, M. D., Perez-Belis, V., 2018. Identifying design guidelines to meet the circular economy principles: A case study on electric and electronic equipment. *Journal of Environmental Management, vol. 228*, pp. 483-494. doi: 10.1016/j.jenvman.2018.08.014

Brezet, H., van Hemel, C., 1997. *Ecodesign: a promising approach to sustainable production and consumption*. Paris, France: UNEP.

Brundtland, G., 1987. Our Common Future—Call for Action. *Environmental Conservation, vol. 14*(4), pp. 291-294. doi: 10.1017/S0376892900016805

Carson, R., 1962. *Silent spring*. New York, United States: Penguin.

Ceschin, F., Gaziulusoy, I., 2016. Evolution of design for sustainability: From product design to design for system innovations and transitions. *Design Studies, vol. 47*, pp. 118-163. doi: 10.1016/j.destud.2016.09.002

Chen, X., Despeisse, M., Johansson, B., 2020. Environmental Sustainability of Digitalization in Manufacturing: A Review. *Sustainability, vol. 12*(24). doi: 10.3390/su122410298

Chun, Y.-Y., Lee, K.-M., 2017. Environmental impacts of the rental business model compared to the conventional business model: a Korean case of water purifier for home use. *The International Journal of Life Cycle Assessment, vol. 22*(7), pp. 1096–1108. doi: 10.1007/s11367-016-1227-1

Churchman, C. W., 1967. *The Systems Approach*. New York, USA: Delacorte Press.

Costa, D., Quinteiro, P., Dias, A. C., 2019. A systematic review of life cycle sustainability assessment: Current state, methodological challenges, and implementation issues. *Sci Total Environ, vol. 686*, pp. 774-787. doi: 10.1016/j.scitotenv.2019.05.435

Cramer, J., 2014. *Milieu: Elementaire Deeltjes: 16*. Amsterdam, The Netherlands: Amsterdam University Press.

Cristóbal, J., Castellani, V., Manfredi, S., Sala, S., 2018. Prioritizing and optimizing sustainable measures for food waste prevention and management. *Waste Manag, vol. 72*, pp. 3-16. doi: 10.1016/j.wasman.2017.11.007

Dawes, J., Bowerman, R., Trepleton, R., 2015. Introduction to the Additive Manufacturing Powder Metallurgy Supply Chain. *Johnson Matthey Technology Review, vol. 59*(3), pp. 243-256. doi: 10.1595/205651315x688686

De Angelis, R., 2018. *Business Models In The Circular Economy - Concepts, Examples and Theory*. Cham, Switzerland: Palgrave Pivot.

de Brito, M. P., Dekker, R., 2003. *A Framework for Reverse Logistics* (ERS-2003-045-LIS). Rotterdam, The Netherlands: Erasmus Universiteit Rotterdam.

Diener, D. L., Williander, M., Tillman, A.-M., 2015. Product-Service-Systems for Heavy-Duty Vehicles – An Accessible Solution to Material Efficiency Improvements? *Procedia CIRP, vol. 30*, pp. 269-274. doi: 10.1016/j.procir.2015.02.027

Earles, J. M., Halog, A., 2011. Consequential life cycle assessment: a review. *The International Journal of Life Cycle Assessment, vol. 16*(5), pp. 445-453. doi: 10.1007/s11367-011-0275-9

EC. 1975. Waste directive 75/442/EEC. *Official Journal of the EU*, 194, pp. 39-41.

EC, 2008. *L 312, 19.11.2008. Directive 2008/98/EC of the European Parliament and of the Council of 19 november 2008 on waste and repealing certain directives.* Official Journal of the EU.

EC, 2015. *Closing the loop - An EU action plan for the Circular Economy*. Brussels, Belgium: European Commission.

EC, 2020. *Communication from the Commission to the European Parliament, the Council, The European Economic and Social Committee and the Committee of the Regions: A new Circular Economy Action, Plan For a cleaner and more competitive Europe*. Brussels, Belgium: Official Journal of the EU.

EMF, 2013. *Towards the circular economy - Vol. 1: an economic and business rationale for an accelerated transition*. Cowes, UK: Ellen MacArthur Foundation. https://www.ellenmacarthurfoundation.org/publications/towards-the-circular-economy-vol-1-an-economic-and-business-rationale-for-an-accelerated-transition

EMF, 2015a. *Towards a Circular Economy: Business Rationale for an Accelerated Transition*. Cowes, UK: Ellen MacArthur Foundation.

EMF, 2015b. *Growth Within: A Circular Economy Vision for a Competitive Europe*. Cowes, UK: Ellen MacArthur Foundation.

Faludi, J., Baumers, M., Maskery, I., Hague, R., 2017. Environmental impacts of selective laser melting: Do printer, powder or power dominate? *Journal of Industrial Ecology, vol. 21, nbr. S1, pp. S144-S156., vol. 21*(S1), pp. S144-S156. doi: 10.1111/jiec.12528

Flyvbjerg, B., 2006. Five Misunderstandings About Case-Study Research. *Qualitative Inquiry, vol. 12*(2), pp. 219-245. doi: 10.1177/1077800405284363

Ford, S., Despeisse, M., 2016. Additive manufacturing and sustainability: an exploratory study of the advantages and challenges. *Journal of Cleaner Production, vol. 137*, pp. 1573-1587. doi: 10.1016/j.jclepro.2016.04.150

Gehin, A., Zwolinski, P., Brissaud, D., 2008. A tool to implement sustainable end-of-life strategies in the product development phase. *Journal of Cleaner Production, vol. 16*(5), pp. 566-576. doi: 10.1016/j.jclepro.2007.02.012

Geissdoerfer, M., Morioka, S. N., de Carvalho, M. M., Evans, S., 2018a. Business models and supply chains for the circular economy. *Journal of Cleaner Production, vol. 190*, pp. 712-721. doi: 10.1016/j.jclepro.2018.04.159

Geissdoerfer, M., Pieroni, M. P. P., Pigosso, D. C. A., Soufani, K., 2020. Circular business models: A review. *Journal of Cleaner Production, vol. 277*. doi: 10.1016/j.jclepro.2020.123741

Geissdoerfer, M., Savaget, P., Bocken, N. M. P., Hultink, E. J., 2017. The Circular Economy – A new sustainability paradigm? *Journal of Cleaner Production, vol. 143*, pp. 757-768. doi: 10.1016/j.jclepro.2016.12.048

Geissdoerfer, M., Vladimirova, D., Evans, S., 2018b. Sustainable business model innovation: A review. *Journal of Cleaner Production, vol. 198*, pp. 401-416. doi: 10.1016/j.jclepro.2018.06.240

Ghisellini, P., Cialani, C., Ulgiati, S., 2016. A review on circular economy: the expected transition to a balanced interplay of environmental and economic systems. *Journal of Cleaner Production, vol. 114*, pp. 11-32. doi: 10.1016/j.jclepro.2015.09.007

Gibbert, M., Ruigrok, W., Wicki, B., 2008. What passes as a rigorous case study? *Strategic Management Journal, vol. 29*(13), pp. 1465-1474. doi: 10.1002/smj.722

Graedel, T. E., Allenby, B. R., 2010. *Industrial ecology and sustainable engineering*. New Jersey, US: Prentice Hall.

Graedel, T. E., Allwood, J., et al., 2011. What Do We Know About Metal Recycling Rates? *Journal of Industrial Ecology, vol. 15*(3), pp. 355-366. doi: 10.1111/j.1530-9290.2011.00342.x

Guinée, J., van den Bergh, J., et al., 1999. Evaluation of risks of metal flows and accumulation in economy and environment. *Ecological Economics, vol. 30*(1), pp. 47-65. doi: 10.1016/s0921-8009(98)00069-x

Gutowski, T., Jiang, S., et al., 2017. Note on the Rate and Energy Efficiency Limits for Additive Manufacturing. *Journal of Industrial Ecology, vol. 21*(S1), pp. S69-S79. doi: 10.1111/jiec.12664

Haupt, M., Zschokke, M., 2017. How can LCA support the circular economy?—63rd discussion forum on life cycle assessment, Zurich, Switzerland, November 30, 2016. *The International Journal of Life Cycle Assessment, vol. 22*(5), pp. 832-837. doi: 10.1007/s11367-017-1267-1

Hauschild, M. Z., Potting, J., 2005. *Spatial Differentiation in Life Cycle Impact Assessment - the EDIP2003 Methodology*. Copenhagen: Environmental News No. 80. The Danish Ministry of the Environment - Environmental Protection Agency.

Hoffmann, B. S., de Simone Morais, J., Teodoro, P. F., 2020. Life cycle assessment of innovative circular business models for modern cloth diapers. *Journal of Cleaner Production, vol. 249*. doi: 10.1016/j.jclepro.2019.119364

Huang, R., Riddle, M. E., Graziano, D., Das, S., Nimbalkar, S., Cresko, J., Masanet, E., 2017. Environmental and Economic Implications of Distributed Additive Manufacturing: The Case of Injection Mold Tooling. *Journal of Industrial Ecology, vol. 21*(S1), pp. S130-S143. doi: 10.1111/jiec.12641

Ingelstam, L., 2002. *System - att tänka över samhälle och teknik (Systems - to think over society and technology)*. Kristianstad, Sweden: Swedish Energy Agency.

Ingemarsdotter, E., Jamsin, E., Kortuem, G., Balkenende, R., 2019. Circular Strategies Enabled by the Internet of Things—A Framework and Analysis of Current Practice. *Sustainability, vol. 11*(20). doi: 10.3390/su11205689

IPCC, 2018. *Global Warming of 1.5°C. 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*. Geneva, Switzerland: Intergovernmental Panel on Climate Change (IPCC).

ISO 14040, 2006. Environmental management—life cycle assessment—principles and framework ISO 14040:2006. Geneva, Switzerland: International Organization for Standardization.

Kaddoura, M., Kambanou, M. L., Tillman, A.-M., Sakao, T., 2019. Is Prolonging the Lifetime of Passive Durable Products a Low-Hanging Fruit of a Circular Economy? A Multiple Case Study. *Sustainability, vol. 11*(18). doi: 10.3390/su11184819

Kamps, T., Lutter-Guenther, M., Seidel, C., Gutowski, T., Reinhart, G., 2018. Cost- and energy-efficient manufacture of gears by laser beam melting. *CIRP Journal of Manufacturing Science and Technology, vol. 21*, pp. 47-60. doi: 10.1016/j.cirpj.2018.01.002

Kellens, K., Baumers, M., Gutowski, T. G., Flanagan, W., Lifset, R., Duflou, J. R., 2017. Environmental dimensions of additive manufacturing: Mapping application domains and their environmental implications. *Journal of Industrial Ecology, vol. 21*(S1), pp. S49-S68. doi: 10.1111/jiec.12629

Kerdlap, P., Gheewala, S. H., Ramakrishna, S., 2021. To Rent or Not to Rent: A Question of Circular Prams from a Life Cycle Perspective. *Sustain Prod Consum, vol. 26*, pp. 331-342. doi: 10.1016/j.spc.2020.10.008

Kirchherr, J., Reike, D., Hekkert, M., 2017. Conceptualizing the circular economy: An analysis of 114 definitions. *Resources, Conservation and Recycling, vol. 127*, pp. 221-232. doi: 10.1016/j.resconrec.2017.09.005

Kjaer, L. L., Pigosso, D. C. A., Niero, M., Bech, N. M., McAloone, T. C., 2018. Product/Service-Systems for a Circular Economy: The Route to Decoupling Economic Growth from Resource Consumption? *Journal of Industrial Ecology*. doi: 10.1111/jiec.12747

Kravchenko, M., Pigosso, D. C. A., McAloone, T. C., 2019. Towards the ex-ante sustainability screening of circular economy initiatives in manufacturing companies: Consolidation of leading sustainability-related performance indicators. *Journal of Cleaner Production, vol. 241*. doi: 10.1016/j.jclepro.2019.118318

Lewis, G. M., Buchanan, C. A., et al., 2019. Green Principles for Vehicle Lightweighting. *Environ Sci Technol, vol. 53*(8), pp. 4063-4077. doi: 10.1021/acs.est.8b05897

Lifset, R., 2017. 3D Printing and Industrial Ecology. *Journal of Industrial Ecology, vol. 21*(S1), pp. S6-S8. doi: 10.1111/jiec.12669

Linder, M., Williander, M., 2017. Circular Business Model Innovation: Inherent Uncertainties. *Business Strategy and the Environment, vol. 26*(2), pp. 182-196. doi: 10.1002/bse.1906

Lindkvist, M., Baumann, H., 2017. Analyzing How Governance of Material Efficiency Affects the Environmental Performance of Product Flows: A Comparison of Product Chain Organization of Swedish and Dutch Metal Packaging Flows. *Recycling, vol. 2*(4). doi: 10.3390/recycling2040023

Liu, Z., Jiang, Q., Cong, W., Li, T., Zhang, H.-C., 2018. Comparative study for environmental performances of traditional manufacturing and directed energy deposition processes. *International Journal of Environmental Science and Technology, vol. 15*(11), pp. 2273-2282. doi: 10.1007/s13762-017-1622-6

Ljunggren Söderman, M., André, H., 2019. Effects of circular measures on scarce metals in complex products – Case studies of electrical and electronic equipment. *Resources, Conservation and Recycling, vol. 151*, pp. 104464. doi: https://doi.org/10.1016/j.resconrec.2019.104464

Löfgren, B., 2012. *The green engineer as an enabler of life-cycle management in manufacturing: models and practices.* (PhD), Gothenburg, Sweden: Chalmers University of Technology.

Louvis, E., Fox, P., Sutcliffe, C. J., 2011. Selective laser melting of aluminium components. *Journal of Materials Processing Technology, vol. 211*(2), pp. 275-284. doi: 10.1016/j.jmatprotec.2010.09.019

Lüdeke-Freund, F., Gold, S., Bocken, N. M. P., 2018. A Review and Typology of Circular Economy Business Model Patterns. *Journal of Industrial Ecology, vol. 23*(1), pp. 36-61. doi: 10.1111/jiec.12763

Luttropp, C., Brohammer, G., 2014. *EcoDesign Roadmap* (1 ed.). Lund, Sweden: Studentlitteratur.

Magretta, J., 2002. Why business models matter. *Harvard Business Review, vol. 80(5)*, pp. 86-92.

Mami, F., Revéret, J.-P., Fallaha, S., Margni, M., 2017. Evaluating Eco-Efficiency of 3D Printing in the Aeronautic Industry. *Journal of Industrial Ecology, vol. 21*(S1), pp. S37-S48. doi: 10.1111/jiec.12693

Manfredi, S., Pant, R., Pennington, D. W., Versmann, A., 2011. Supporting environmentally sound decisions for waste management with LCT and LCA. *The International Journal of Life Cycle Assessment, vol. 16*(9), pp. 937-939. doi: 10.1007/s11367-011-0315-5

Martin, M., Heiska, M., Björklund, A., 2021. Environmental assessment of a product-service system for renting electric-powered tools. *Journal of Cleaner Production, vol. 281*. doi: 10.1016/j.jclepro.2020.125245

Massa, L., Tucci, C. L., 2014. Business Model Innovation. In The Oxford handbook of innovation management, ed. M. Dodgson, D. M. Gann, & N. Phillips. Oxford, United Kingdom: Oxford University Press.

Meadows, D., Meadows, D., Randers, J., Behrens III, W., 1972. *Limits to growth* (Vol. 102). New York: Universe Books.

Mendoza, J. M. F., D'Aponte, F., Gualtieri, D., Azapagic, A., 2019. Disposable baby diapers: Life cycle costs, eco-efficiency and circular economy. *Journal of Cleaner Production, vol. 211*, pp. 455-467. doi: 10.1016/j.jclepro.2018.11.146

Mistra REES, 2020. Mistra REES – Resource-Efficient and Effective Solutions based on circular economy thinking. http://mistrarees.se/. (2020-12-29)

Mognol, P., Lepicart, P., Perry, N., 2006. Rapid prototyping: energy and environment in the spotlight. *Rapid Prototyping Journal, vol. 12*(1), pp. 26-34. doi: 10.1108/13552540610637246

Mont, O., 2002. Clarifying the concept of product-service system. *Journal of Cleaner Production, vol. 10*, pp. 237-245. doi: 10.1016/S0959-6526(01)00039-7

Moreno, M., De los Rios, C., Rowe, Z., Charnley, F., 2016. A Conceptual Framework for Circular Design. *Sustainability, vol. 8*(9). doi: 10.3390/su8090937

Moylan, S., Whitenton, E., Lane, B., Slotwinski, J., 2014. *Infrared thermography for laser-based powder bed fusion additive manufacturing processes*. 40th Annual Review of Progress in Quantitative Nondestructive Evaluation. AIP Conf. Proc. 1581, pp. 1191-1196. doi: 10.1063/1.4864956.

Nilsson-Lindén, H., Rosén, M., Baumann, H., 2019. Product chain collaboration for sustainability: A business case for life cycle management. *Business Strategy and the Environment, vol. 28*(8), pp. 1619-1631. doi: 10.1002/bse.2388

Norris, G., 2001. Integrating Life Cycle Cost Analysis and LCA. *International Journal of Life Cycle Assessment, vol. 6*(2), pp. 118-120. doi: 10.1007/BF02977849

Nußholz, J. L. K., 2020. *Circular Business Model Design: Business Opportunities from Retaining Value of Products and Materials.* (PhD thesis), Lund, Sweden: International Institute for Industrial Environmental Economics, Lund University.

Osterwalder, A., Pigneur, Y., 2010. *Business Model Generation: A Handbook for Visionaries, Game Changers, and Challengers*. Hoboken, New Jersey: John Wiley & Sons.

Packard, V., 1960. *The Waste Makers. A Startling Revelation of Planned Wastefulness and Obsolescence in Industry Today*. New York, United States: David McKay Company.

Piekarski, C. M., da Luz, L. M., Zocche, L., de Francisco, A. C., 2013. Life Cycle Assessment as Entrepreneurial Tool for Business Management and Green Innovations. *Journal of Technology Management & Innovation, vol. 8*(1), pp. 44-53. doi: 10.4067/S0718-27242013000100005

Pieroni, M. P. P., McAloone, T. C., Pigosso, D. C. A., 2020. From theory to practice: systematising and testing business model archetypes for circular economy. *Resources, Conservation and Recycling, vol. 162*. doi: 10.1016/j.resconrec.2020.105029

Pigosso, D. C. A., McAloone, T. C., Rozenfeld, H., 2015. Characterization of the State-of-the-art and Identification of Main Trends for Ecodesign Tools and Methods: Classifying Three Decades of Research and Implementation. *Journal of the Indian Institute of Science*, 95(4), pp. 405-427.

Pizzol, M., Laurent, A., Sala, S., Weidema, B., Verones, F., Koffler, C., 2017. Normalisation and weighting in life cycle assessment: quo vadis? *The International Journal of Life Cycle Assessment, vol. 22*(6), pp. 853-866. doi: 10.1007/s11367-016-1199-1

Potting, J., Hekkert, M., Worrell, E., Hanemaaijer, A., 2017. *Circular Economy: Measuring Innovation in the Product Chain*. The Hague, The Netherlands: PBL Netherlands Environmental Assessment Agency.

Priarone, P. C., Ingarao, G., di Lorenzo, R., Settineri, L., 2017. Influence of Material-Related Aspects of Additive and Subtractive Ti-6Al-4V Manufacturing on Energy Demand and Carbon Dioxide Emissions. *Journal of Industrial Ecology, vol. 21*(S1), pp. S191-S202. doi: 10.1111/jiec.12523

Rashid, A., Asif, F. M. A., Krajnik, P., Nicolescu, C. M., 2013. Resource Conservative Manufacturing: an essential change in business and technology paradigm for sustainable manufacturing. *Journal of Cleaner Production, vol. 57*, pp. 166-177. doi: 10.1016/j.jclepro.2013.06.012

Reike, D., Vermeulen, W. J. V., Witjes, S., 2018. The circular economy: New or Refurbished as CE 3.0? — Exploring Controversies in the Conceptualization of the Circular Economy through a Focus on History and Resource Value Retention Options. *Resources, Conservation and Recycling, vol. 135*, pp. 246-264. doi: 10.1016/j.resconrec.2017.08.027

Reuter, B., 2016. Assessment of sustainability issues for the selection of materials and technologies during product design: a case study of lithium-ion batteries for electric vehicles. *International Journal on Interactive Design and Manufacturing, vol. 10*(3), pp. 217-227. doi: 10.1007/s12008-016-0329-0

Rockström, J., Steffen, W., et al., 2009. Planetary Boundaries: Exploring the Safe Operating Space for Humanity. *Ecology and Society*, 14(2).

Rombouts, M., Kruth, J. P., Froyen, L., Mercelis, P., 2006. Fundamentals of Selective Laser Melting of alloyed steel powders. *CIRP Annals, vol. 55*(1), pp. 187-192. doi: 10.1016/s0007-8506(07)60395-3

Rosa, P., Sassanelli, C., Terzi, S., 2019. Towards Circular Business Models: A systematic literature review on classification frameworks and archetypes. *Journal of Cleaner Production, vol. 236*. doi: 10.1016/j.jclepro.2019.117696

Rossi, M., Germani, M., Zamagni, A., 2016. Review of ecodesign methods and tools. Barriers and strategies for an effective implementation in industrial companies. *Journal of Cleaner Production, vol. 129*, pp. 361-373. doi: 10.1016/j.jclepro.2016.04.051

Saade, M. R. M., Yahia, A., Amor, B., 2020. How has LCA been applied to 3D printing? A systematic literature review and recommendations for future studies. *Journal of Cleaner Production, vol. 244*. doi: 10.1016/j.jclepro.2019.118803

Sandén, B. A., 2004. Technology path assessment for sustainable technology development. *Innovation, vol. 6*(2), pp. 316-330. doi: 10.5172/impp.2004.6.2.316

Sarasini, S., Linder, M., 2018. Integrating a business model perspective into transition theory: The example of new mobility services. *Environmental Innovation and Societal Transitions, vol. 27*, pp. 16-31. doi: 10.1016/j.eist.2017.09.004

Schaltegger, S., Lüdeke-Freund, F., Hansen, E. G., 2012. Business cases for sustainability: the role of business model innovation for corporate sustainability. *International Journal of Innovation and Sustainable Development, vol. 6*(2), pp. 95-199. doi: 10.1177/1086026615599806

Schau, E. M., Traverso, M., Finkbeiner, M., 2012. Life cycle approach to sustainability assessment: a case study of remanufactured alternators. *Journal of Remanufacturing, vol. 2*(5), pp. 1-14. doi: 10.1186/2210-4690-2-5

Schöggl, J.-P., Stumpf, L., Baumgartner, R. J., 2020. The narrative of sustainability and circular economy - A longitudinal review of two decades of research. *Resources, Conservation and Recycling, vol. 163*. doi: 10.1016/j.resconrec.2020.105073

Siggelkow, N., 2007. Persuasion With Case Studies. *Academy of Management Journal, vol. 50*(1). doi: 10.5465/amj.2007.24160882

Sihvonen, S., Ritola, T., 2015. Conceptualizing ReX for Aggregating End-of-life Strategies in Product Development. *Procedia CIRP, vol. 29*, pp. 639-644. doi: 10.1016/j.procir.2015.01.026

Soo, V. K., Compston, P., Doolan, M., 2016. Is the Australian Automotive Recycling Industry Heading towards a Global Circular Economy? – A Case Study on Vehicle Doors. *Procedia CIRP, vol. 48*, pp. 10-15. doi: 10.1016/j.procir.2016.03.099

Stahel, W. R., 2010. *The performance economy* (2nd ed.). London, U.K.: Palgrave Macmillan.

Stahel, W. R., Clift, R., 2016. Stocks and Flows in the Performance Economy. In Taking Stock of Industrial Ecology, ed. R. Clift & A. Druckman, (pp. 137-158). Cham: Springer International Publishing.

Steen, B., 2015. *The EPS 2015 impact assessment method – An overview* (2015:5). Gothenburg, Sweden: Swedish Life Cycle Center.

Sundin, E., 2004. *Product and Process Design for Successful Remanufacturing.* (PhD dissertation), Linköping, Sweden: Linköping University.

Sundin, E., 2009. Life-Cycle Perspectives of Product/Service-Systems: In Design Theory. In Introduction to Product/Service-System Design, ed. T. Sakao & M. Lindahl, (pp. 31-49).

Tillman, A.-M., 2000. Significance of decision-making for LCA methodology. *Environmental Impact Assessment Review, vol. 20*, pp. 113-123. doi: 10.1016/S0195-9255(99)00035-9

Tillman, A.-M., 2012. New ways of industry-academia interaction, for new solutions on sustainability. *Presentation at the Inauguration of the SKF-Chalmers University of Technology Centre for Sustainability*, Gothenburg, 27 March, 2012.

Tillman, A.-M., Ljunggren Södergren, M., André, H., Böckin, D., Willskytt, S., 2020. *Circular economy and its impact on use of natural resources and the environment: Chapter from the upcoming book "Resource-Efficient and Effective Solutions – A handbook on how to develop and provide them"* (2020:1). Gothenburg, Sweden: Chalmers University of Technology.

Tillman, A.-M., Svingby, M., Lundström, H., 1998. Life Cycle Assessment of Municipal Waste Water Systems. *International Journal of Life Cycle Assessment, vol. 3*(3), pp. 145-157. doi: 10.1007/BF02978823

Tschiggerl, K., Sledz, C., Topic, M., 2018. Considering environmental impacts of energy storage technologies: A life cycle assessment of power-to-gas business models. *Energy, vol. 160*, pp. 1091-1100. doi: 10.1016/j.energy.2018.07.105

Tukker, A., 2015. Product services for a resource-efficient and circular economy–a review. *Journal of Cleaner Production, vol. 97*, pp. 76-91. doi: 10.1016/j.jclepro.2013.11.049

Tukker, A., Tischner, U., 2006. Product-services as a research field: past, present and future. Reflections from a decade of research. *Journal of Cleaner Production, vol. 14*(17), pp. 1552-1556. doi: 10.1016/j.jclepro.2006.01.022

UNEP, 2010. *ABC of SCP. Clarifying Concepts on Sustainable Consumption and Production*. United Nations Environment Programme.

UNEP, 2017. *Resource Efficiency: Potential and Economic Implications - A Report of the International Resource Panel*. United Nations International Resource Panel.

Urbinati, A., Chiaroni, D., Chiesa, V., 2017. Towards a new taxonomy of circular economy business models. *Journal of Cleaner Production, vol. 168*, pp. 487-498. doi: 10.1016/j.jclepro.2017.09.047

van Berkel, R., Willems, E., Lafleur, M., 1997. The Relationship between Cleaner Production and Industrial Ecology. *Journal of Industrial Ecology, vol. 1*(1), pp. 51-66. doi: 10.1162/jiec.1997.1.1.51

Vezzoli, C., Manzini, E., 2008. *Design For Environmental Sustainability*. London, United Kingdom: Springer.

Villares, M., Işıldar, A., van der Giesen, C., Guinée, J., 2017. Does ex ante application enhance the usefulness of LCA? A case study on an emerging technology for metal recovery from e-waste. *The International Journal of Life Cycle Assessment, vol. 22*(10), pp. 1618-1633. doi: 10.1007/s11367-017-1270-6

Volvo Group, 2017. Metal 3d printing: technology of the future for lighter and more compact engines [Press release]. Retrieved from https://corporate.renault-trucks.com/en/press-releases/metal-3d-printing-technology-of-the-future-for-lighter-and-more-compact-engines.html (2020-12-14)

von Bertalanffy, L., 1969. *General System Theory: Foundations, Development, Applications*. New York, New York, USA: George Braziller, Inc. ISBN: 978-0807604533.

Walachowicz, F., Bernsdorf, I., et al., 2017. Comparative Energy, Resource and Recycling Lifecycle Analysis of the Industrial Repair Process of Gas Turbine Burners Using Conventional Machining and Additive Manufacturing. *Journal of Industrial Ecology, vol. 21*(S1), pp. S203-S215. doi: 10.1111/jiec.12637

Whalen, K. A., 2019. Three circular business models that extend product value and their contribution to resource efficiency. *Journal of Cleaner Production, vol. 226*, pp. 1128-1137. doi: 10.1016/j.jclepro.2019.03.128

Willskytt, S., 2020. *Resource efficient products in a circular economy – The case of consumables - From environmental and resource assessment to design guidelines.* (PhD), Gothenburg, Sweden: Chalmers University of Technology.

Willskytt, S., Brambila-Macias, S. A., 2020. Design Guidelines Developed from Environmental Assessments: A Design Tool for Resource-Efficient Products. *Sustainability, vol. 12*(12). doi: 10.3390/su12124953

Wohler's Associates, 2016. *3D Printing and Additive Manufacturing State of the Industry: Annual Worldwide Progress Report*. Fort Collins, CO, USA: Wohler's Associates.

Yan, J., Feng, C., 2013. Sustainable design-oriented product modularity combined with 6R concept: a case study of rotor laboratory bench. Clean Technol. Environ. Policy vol. 16(1), pp. 95–109.

Yin, R. K., 1981. The case study as a serious research strategy. *Knowledge*, 3(1), pp. 97-114.

Yong, R., 2007. The circular economy in China. *Journal of Material Cycles and Waste Management, vol. 9*(2), pp. 121-129. doi: 10.1007/s10163-007-0183-z

Yule, A. J., Dunkley, J. J., 1994. *Atomization of melts for powder production and spray deposition*. New York: Oxford University Press.

Zamani, B., Sandin, G., Peters, G. M., 2017. Life cycle assessment of clothing libraries: can collaborative consumption reduce the environmental impact of fast fashion? *Journal of Cleaner Production, vol. 162*, pp. 1368-1375. doi: 10.1016/j.jclepro.2017.06.128

Zhang, W., Guo, J., Gu, F., Gu, X., 2018. Coupling life cycle assessment and life cycle costing as an evaluation tool for developing product service system of high energy-consuming equipment. *Journal of Cleaner Production, vol. 183*, pp. 1043-1053. doi: 10.1016/j.jclepro.2018.02.146

Zhijun, F., Nailing, Y., 2007. Putting a circular economy into practice in China. *Sustainability Science, vol. 2*(1), pp. 95-101. doi: 10.1007/s11625-006-0018-1

Zott, C., Amit, R., Massa, L., 2011. The Business Model: Recent Developments and Future Research. *Journal of Management, vol. 37*(4), pp. 1019-1042. doi: 10.1177/0149206311406265