# Biomass residues as resources

## An expanded life-cycle perspective

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## An expanded life-cycle perspective

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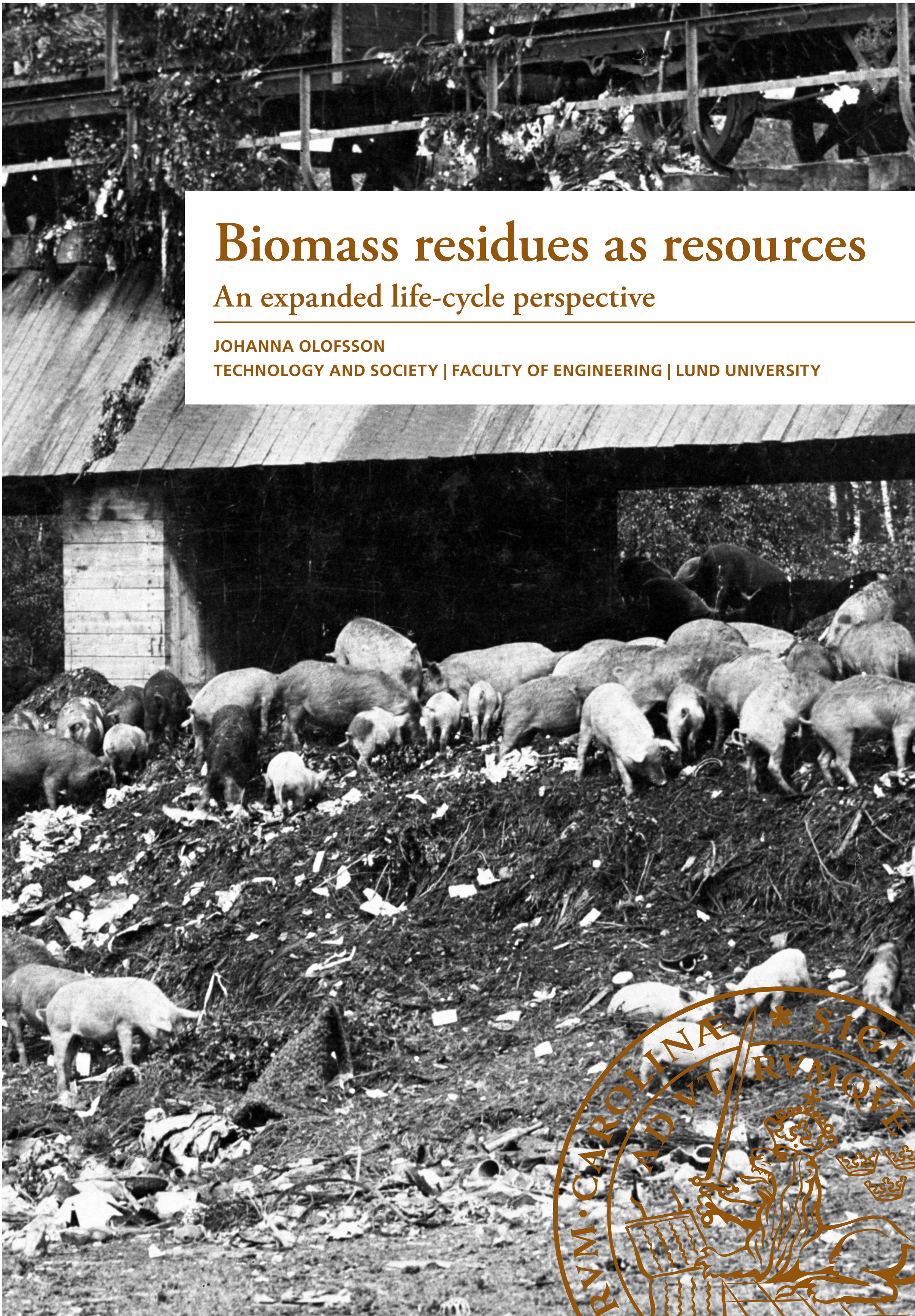

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Different ways of managing biomass residues, for instance as resources for bio-based products, are associated with different opportunities for climate-change mitigation as well as risks. This thesis investigates and discusses how some important aspects of biomass valorisation processes, and of the method of life cycle assessment, influence the conclusions for what appears to be a better use of biomass residues from a climate-change mitigation perspective.

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Biomass residues as resources – An expanded life-cycle perspective

# Biomass residues as resources

## An expanded life-cycle perspective

Johanna Olofsson

LUND UNIVERSITY

DOCTORAL DISSERTATION

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**Abstract:**

Biomass residues have been identified as potentially promising resources for production of bio-based products and fuels with low climate impacts. Unlike primary biomass such as crops, using residual biomass may avoid issues such as competition over land for food and feed production. To assess the climate impacts of products made from residual biomass, a life cycle assessment (LCA) may be used, but its results are known to be sensitive to method choices. The aim of this thesis, therefore, is to better understand and illustrate how different assessment approaches affect conclusions regarding the climate impacts of different management alternatives for biomass residues.

Different factors affect either the greenhouse gas emissions related to valorisation of biomass residues, or the climate impacts of those emissions. These factors range from the design of valorisation processes, including the use of enzymes and energy carriers in production, to the way LCA is applied, including how the upstream processes that the residues come from are considered, and the way in which $CO_2$ fluxes from biomass are considered in the assessment. Finally, the method used for climate impact assessment can affect the conclusions regarding the climate-change mitigation potential, especially for forest residual biomass. Whether valorisation of biomass residues provides climate benefits therefore depends on how the bio-based products are produced, what they are compared to and when, and on the specific goal of climate-change mitigation.

Valorisation strategies for residual biomass should increasingly consider the upstream processes of biomass residues, as these materials are increasingly considered valuable. When these processes are included in LCA, they can have a significant impact on the conclusions drawn in some cases of residual biomass valorisation. It is also essential to consider other valuable uses of residual biomass materials to sustain long-term productivity and sustainability of primary production systems. Both these strategies are important in circular bioeconomies, but available circularity assessment methods for bio-based products primarily focus on the former, and still fail to consider processes and inputs related to high climate impacts.

The idea of valorising residual biomass into products with low climate impacts is thus more complex than at first sight, and many parameters can affect the conclusions. A better understanding of this complexity can potentially lead to a more nuanced understanding of the possibilities and risks related to using biomass residues as resources.

**Key words:** Biomass residues, biofuels, life cycle assessment, climate change, circular bioeconomy

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*To my family*

# Table of Contents

# Abstract

Biomass residues have been identified as potentially promising resources for production of bio-based products and fuels with low climate impacts. Unlike primary biomass such as crops, using residual biomass may avoid issues such as competition over land for food and feed production. To assess the climate impacts of products made from residual biomass, a life cycle assessment (LCA) may be used, but its results are known to be sensitive to method choices. The aim of this thesis, therefore, is to better understand and illustrate how different assessment approaches affect conclusions regarding the climate impacts of different management alternatives for biomass residues.

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The idea of valorising residual biomass into products with low climate impacts is thus more complex than at first sight, and many parameters can affect the conclusions. A better understanding of this complexity can potentially lead to a more nuanced understanding of the possibilities and risks related to using biomass residues as resources.

# Populärvetenskaplig sammanfattning

Organiska restprodukter som halm från jordbruket, grenar från skogsindustrin, matrester, slaktrester och liknande skulle kunna ersätta fossila råvaror i tillverkning av bränslen och andra produkter. I den här avhandlingen undersöker jag hur olika faktorer påverkar hur bra det verkar vara ur klimatsynpunkt att utnyttja organiska restprodukter som råvaror. Jag visar att det kan vara bra ur klimatsynpunkt att utnyttja organiska restprodukter men också att frågan är mer komplex än så och att det kan finnas flera olika svar. Vad som verkar vara ett bra val beror i vissa fall på när det är viktigt att minska klimatpåverkan. Vi behöver också fråga oss hur vi vill att våra samhällen, odlingsmetoder och skogsbruk ska se ut, eftersom restprodukterna vi möjligen kan utnyttja är tätt sammankopplade med andra produkter.

Idén att använda organiska restprodukter istället för odlade grödor och träd kommer delvis från att de inte konkurrerar om landyta på samma sätt. Produkter som tillverkats från odlade grödor riskerar att konkurrera med t.ex. matproduktion och är av den anledningen kanske inte bättre ur klimatsynpunkt än produkter som tillverkats med fossila råvaror som olja, kol och naturgas.

För att förstå om det kan bli bättre ur klimatsynpunkt att tillverka produkter av organiska restprodukter använder jag livscykelanalys (LCA). LCA-metoden används i många sammanhang för att beräkna produkters klimat- och miljöpåverkan, men de resultat man får kan bli väldigt olika beroende av olika val som görs inom metoden. Här kan man se på organiska restprodukter på två sätt: dels som en råvara för produkter, dels som en rest som måste hanteras på ett eller annat sätt. Utifrån dessa perspektiv visar jag hur olika faktorer kan påverka slutsatserna för organiska restprodukter.

Bland annat spelar det roll hur restprodukterna används och till vad, och till exempel vilken energi och andra medel som krävs. Det spelar också roll hur restproduktens livscykel utformas inom LCA-metoden, och hur dess klimatpåverkan beräknas. När restprodukter används som värdefulla råvaror är det också motiverat att ta hänsyn till var de kommer ifrån. Generellt får restprodukter från processer och system med en hög klimatpåverkan också en högre klimatpåverkan i beräkningar.

Om man ser till vad restprodukter annars kan användas till, så är det inte alltid man får större klimatnytta av att utnyttja restprodukter än att låta dem vara, till exempel i skogen. Om restprodukter som halm och grenar får ligga kvar kan de bidra med viktiga näringsämnen till jorden, men metoder för att utvärdera den typen av kretslopp behöver utvecklas. Om det blir bättre ur klimatsynpunkt att utnyttja organiska restprodukter är alltså olika från fall till fall, och kan bero på våra prioriteringar och hur andra, större system som jordbruk, skogsbruk och energiproduktion utvecklas.

# List of publications

This thesis is based on the following publications, referred to by their Roman numerals:

I. Olofsson, J., Barta, Z., Börjesson, P. and Wallberg, O. (2017) Integrating enzyme fermentation in lignocellulosic ethanol production: life-cycle assessment and techno-economic analysis. Biotechnology for Biofuels 10:51. Available at: http://dx.doi.org/10.1186/s13068-017-0733-0

II. Olofsson, J. and Börjesson, P. (2018) Residual biomass as resource – Life-cycle environmental impact of wastes in circular resource systems. Journal of Cleaner Production 196, pp. 997-1006. Available at: https://doi.org/10.1016/j.jclepro.2018.06.115

III. Olofsson, J. (2021) Time-Dependent Climate Impact of Utilizing Residual Biomass for Biofuels—The Combined Influence of Modelling Choices and Climate Impact Metrics. Energies 14(14): 4219. Available at: https://doi.org/10.3390/en14144219

IV. Olofsson, J. Circularity for biomass residues: assessing indicators and climate impacts. Under review.

# Author's contribution to the publications

I. I designed the environmental assessment in collaboration with Pål Börjesson while Zsolt Barta and Ola Wallberg were responsible for the modelling, the techno-economic analysis and original idea. I conducted most of the data collection and calculations for the environmental assessment and wrote most of the connected parts of the paper.

II. I designed the study in collaboration with Pål Börjesson. I conducted most of the data collection and analysis and wrote most of the paper.

III. I conducted the study and wrote the article as sole author.

IV. I conducted the study and wrote the article as sole author.

# Other relevant publications

Pettersson, M., Olofsson, J., Björnsson, L. and Börjesson, P. (2022) Reductions in greenhouse gas emissions through innovative co-production of bio-oil in combined heat and power plants. *Applied Energy* 324:119637.

Brekke, A., Lyng, K.-A., Olofsson, J. and Szulecka, J. (2019) Life cycle assessment – A governance tool for transition towards a circular bioeconomy? In Klitkou, A., Fevolden, A.M., and Capasso, M. (Eds.). From Waste to Value: Valorisation Pathways for Organic Waste Streams in Circular Bioeconomies (1st ed.). Routledge.

Olofsson, J. and Börjesson, P. (2018) Greenhouse gas emissions of methanol from co-gasification of black liquor with by-product biomass. Report 107, Environmental and Energy Systems Studies, LTH, Lund University, Lund, Sweden.

Held, J. and Olofsson, J. (2018) LignoSys - System study of small scale thermochemical conversion of lignocellulosic feedstock to biomethane. Report 008:2018, Renewable Energy Technology International AB, Lund, Sweden.

Olofsson, J. and Börjesson, P. (2016) Nedlagd åkermark för biomassaproduktion – Kartläggning och potentialuppskattning. Report 2016:01, f3 The Swedish Knowledge Centre for Renewable Transportation Fuels and Foundation, Sweden.

# Acknowledgements

I am very happy and immensely grateful to not have been alone in doing this PhD. The words on this page fail to cover the entirety of it, but I will try to provide some highlights.

First, I would like to thank my supervisors, Pål and Lovisa, for your trust and support. Pål, thank you for all the encouragement and for making room for my ideas, including the not-so-streamlined ones. Lovisa, thank you for taking the time to help me out when I needed it, and for lending a sharp eye when mine were blurry.

To all my colleagues at the division, thank you for the countless inspiring discussions in more and less formal settings, and for reading and discussing ideas and drafts. I have greatly appreciated the feeling of being part of something that tries to care and matter, especially when my own work seemed like an ever-expanding rabbit hole.

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I am very grateful to have had the opportunity to be a part of different research projects. Thank you to everyone involved for sharing your perspectives and knowledge, and for allowing me a place to learn.

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# Abbreviations

| | |
|---|---|
| AGTP | Absolute global temperature change potential |
| AGWP | Absolute global warming potential |
| GHG | Greenhouse gas |
| GTP | Global temperature change potential |
| GWP | Global warming potential |
| HVO | Hydrotreated vegetable oils |
| IPCC | Intergovernmental Panel on Climate Change |
| ISO | International Organization for Standardization |
| LCA | Life cycle assessment |
| MCI | Material circularity indicator |
| RED | Renewable energy directive |

# 1 Introduction

Fifteen years ago, two articles were published in an issue of the journal *Science* on the topic of biofuels and their climate impact (Fargione et al., 2008, Searchinger et al., 2008). The authors essentially argued that due to effects from direct and indirect land use change[1], the climate impact of biofuels made from edible biomass is higher than previously thought and for some biofuels, substantially higher than the climate impact of fossil fuels. One article "*highlights the value of biofuels from waste products (26) because they can avoid land-use change and its emissions*" (Searchinger et al., 2008), and the other concludes that "*biofuels made from waste biomass (…) incur little or no carbon debt and can offer immediate and sustained GHG advantages*" (Fargione et al., 2008). Apart from the potential in using abandoned land for bioenergy purposes, the waste biomass mentioned as potential feedstocks for bioenergy in these articles include crop residues or crop waste, slash and thinnings from sustainable forestry, municipal waste and animal waste. In this context, the two articles made reference to another publication each. The first studied harvest of corn stover in the US and found that there was a potential to use the biomass for energy purposes in some regions (Graham et al., 2007). The second investigated biomass potentials in the US and to what extent they could replace the domestic demand for transportation fuel (Perlack et al., 2005). Interestingly, both highlighted limits to harvesting of residues from forestry and agriculture due to the need to preserve soil quality – more specifically, soil organic carbon and future productivity – and neither assessed the climate or environmental impacts associated with valorisation of biomass residues for biofuel production. The articles published in *Science* led to a wide debate on both biofuels and their assessment, but the idea of biomass residues as "safer" feedstocks for bioenergy and bio-based products seemed to catch on as opinion grew against crop-based, or so-called first-generation biofuels.

With the goal of more explicitly assessing the environmental impacts of different parts of production systems, life cycle assessment (LCA) was already an established practice

[1] Direct land use change denotes the change in management of a piece of land when it is used for e.g. cultivation of energy crops. By contrast, indirect land use change is change happening elsewhere as a consequence of the same cultivation.

that had been used both to promote and to criticise biofuels on the merits of their climate and other environmental impacts. During the 21st century, the application of LCA to bioenergy has driven a development of the method (McManus et al., 2015). For instance, it was essentially an expansion of the system boundaries to include land conversion (Fargione et al., 2008), and an addition to the LCA method using economic equilibrium models (Searchinger et al., 2008), which led to the new conclusions for first generation biofuels presented in *Science*. While some aspects of the methods used in the two articles have been disputed, LCA has since continued to be applied to assess climate and other environmental impacts of biofuels and other bio-based products.

As a parallel track, the valorisation of biomass residues is often mentioned as an essential building block of circular bioeconomies (Stegmann et al., 2020). In this context, valorisation implies a process aimed at increasing the economic value of a material. The concept of circular bioeconomies has grown in popularity in both policy (e.g. European Commission, 2018) and research environments (Yaremova et al., 2021). Though the ideas of circular bioeconomies may be ambiguously defined (Tan and Lamers, 2021), the valorisation of biomass residues can be considered a common denominator.

The idea of residues as resources may appear promising, but to confidently state that something has a lesser environmental impact than an alternative, experience should teach us not to settle for appearance. Many LCA studies have indeed studied different cases of residual biomass use as feedstock for fuels and products. Some have indicated 'hotspots' – processes or inputs that contribute a significant share of the total climate impact of a system. For instance, the use of enzymes for conversion of woody biomass was identified as a hotspot for improvement and development (Slade et al., 2009, MacLean and Spatari, 2009). Further studies have shown positive effects from valorisation options since new ways of handling residues may alleviate environmental pressures from unmanaged, or poorly managed, residual biomass (Silalertruksa and Gheewala, 2013, Pfau, 2019). Before the publication of the two articles in *Science*, others had already raised questions about adaptations of the LCA method for assessing biomass residues as resources. For instance, some suggestions for method consideration and improvement have concerned what happens when biomass residues are removed from soils or from other product applications (Wiloso, 2015). For some biomass residues, for instance from forestry and agriculture, one option is to leave the residual biomass on site. Since unharvested biomass residues can then contribute to carbon being sequestered in the soil, and to soil quality and ecosystem functions, assessments of valorisation of such residues should consider the potential reduction of these values (Cherubini et al., 2009, Lal, 2005). These aspects are, however, often dependent on local conditions (Cherubini et al., 2009). Different models and measurements can also provide different bases for assessing the implications of residue removal, for instance concerning the speed of decay of different types of forest biomass residues (Gustavsson et al., 2015), or the contribution to soil carbon storage from straw (Björnsson and Prade, 2021). Other types of biomass residues, for instance from food, feed, or wood processing industries, have to be dealt with, and they are therefore often already utilised or managed for different purposes. If these types of biomass residues are to be considered as feedstocks for new products, their diversion from existing uses should also be considered (Tonini et al., 2016, Pfau, 2015, Wiloso, 2015).

Perhaps on a more theoretical level, a potential shift in the view of residual materials as valuable has also been observed. This could be considered in line with the general idea of circular bioeconomies, but it may also have implications regarding how residual biomass can be treated in LCAs. Pradel et al. (2016) argue that wastewater sludge is in this way increasingly being recognised as a potentially valuable resource for valorisation, and that this may outdate previous assumptions based on the waste status of the sludge, including it being free from any environmental impact arising from the wastewater treatment plant. Similarly, Oldfield et al. (2018) apply two stakeholder perspectives in an LCA study of food and green garden waste, where a circular economy perspective that includes waste valorisation implies a change in the set-up of the study compared to a traditional waste LCA. Both these studies found that the adaptation of the LCA method could affect the conclusions drawn in the studied cases.

Nevertheless, valorisation of residual biomass is, as mentioned, considered a principally important building block of future circular bioeconomies. The assessment of circularity for biomass and residual biomass is receiving increasing attention, but it is also a new practice, with several identified limitations. First, the definition of circularity for biomass is not self-evident. In this context it includes sustainable management and regeneration of primary production systems, efficient use of biomass resources by cascading use and recycling, and closing of nutrient cycles by maintaining biodegradability of materials and avoiding contamination by hazardous substances (Vural Gursel et al., 2022, Navare et al., 2021). Second, many available methods for circularity assessment lack an explicit approach to assess circularity of biomass and bio-based products. For instance, Navare et al. (2021) reviewed 59 circular economy metrics, out of which they assessed three as fully and explicitly considering at least one out of several important aspects of bio-based circularity. These metrics have been applied only to a limited extent and the lack of application to case studies hampers a better understanding of what it is that the methods measure (Jerome et al., 2022).

## 1.1 Research objective and questions

In many ways, the concept and promise of biomass residues as resources appears more complex than at first sight, both in practice and in assessment. LCA is often relied on to provide decision-making processes with information on existing or potential environmental impacts related to different products, processes, and services. For decision-making purposes, seemingly contradictory LCA results and conclusions for the same type of raw materials and products can be considered problematic. On the other hand, even such LCAs can potentially be used to inform decision-making processes and discussions on critical parameters and possible interpretations (Bras-Klapwijk, 1999). With ideas of circular bioeconomies as models for future societies and the basis for new policies, a better understanding is also needed of how and why LCA and circularity assessment may converge or conflict in terms of guiding residual biomass management. It is within this area of developing and clarifying the application of a life-cycle perspective to biomass residues as resources that this thesis aims to contribute.

The aim of this thesis is to better understand and illustrate how different assessment approaches affect conclusions regarding the climate impacts of different management alternatives for biomass residues. This aim departs from the life-cycle perspective and covers comparative set-ups, system boundaries, climate impact assessment methods and other choices made by practitioners, implicitly or explicitly. In addition, the included case studies provide insight into how different method choices matter to specific biomass residues and valorisation options.

Towards this aim, the following research questions are addressed.

- How do different factors influence conclusions and comparisons of the climate impacts of different alternatives of biomass residues' management?
- How can the impacts of primary biomass production and other processes in which biomass residues are created be considered in assessments?
- What implications may such consideration have to conclusions for residual biomass valorisation?
- In what ways do climate impact assessment within LCA and bio-based circularity assessment provide additional or contrasting insights into residual biomass use?

Each of these questions represents an expanded perspective on biomass residues as resources for bio-based products in the sense that they explore different comparative scenarios, widening of system boundaries, and different end goals of climate-change mitigation and circular bioeconomies. In answering the first question, I focus on a limited number of methodological and practical factors that are relevant to biomass residues, including the way in which the life cycles of biomass residues are considered, the design of valorisation processes, and the methods used to calculate climate impacts. This is not to say that there are not other factors that could be of importance; rather, there likely are. Instead, this is a delimitation of the research objective of this thesis. Table 1 illustrates how the research questions are addressed by the four appended papers, and how each paper covers different parts of the life-cycle perspective and cases of residual biomass and bio-based products.

**Table 1. Overview of the papers**
For each paper, the table gives the connections to the research questions, the chapters of this thesis, and an overview of the research approach and case studies. RED: the EU renewable energy directive (2018/2001), ISO: International Organization for Standardization, see chapter 2.

| | **Paper I** | **Paper II** | **Paper III** | **Paper IV** |
|---|---|---|---|---|
| **RQs addressed** | 1 | 2, 3 | 1 | 3, 4 |
| **Findings in chapter** | 3 | 4 | 3, 5 | 4, 5 |
| **LCA assumptions and framings investigated or challenged** | ISO and EU RED methods for biofuels, biogenic $CO_2$ and soil organic carbon inclusion | Zero-burden assumptions, rationales for allocating upstream impacts to residues | Temporal framings of bio-based systems and climate impacts, biogenic $CO_2$ and soil organic carbon inclusion | Circularity for biomass residues as resources, compared to their climate impact |
| **Methods** | LCA, RED, techno-economic assessment | Literature review and analysis | LCA with time-dependent inventories | LCA, circularity assessment |
| **Case studies: biomass residues and bio-based products** | Ethanol from logging residues. Integrated production of enzymes and ethanol | Residues from different food and forest-related processing industries | Logging residues and wheat straw as feedstocks for ethanol and other energy carriers | Tallow for jet-fuel production. Wheat straw for ethanol production |

## 1.2 Scope and delimitations

This thesis combines case studies of specific types of residual biomass and conversion routes with more theoretical arguments about residues' valorisation and the assessment approaches needed to study it. Like biomass in general, biomass residues are not a homogenous group of materials, and they exist in different systems and contexts. It is therefore likely not possible to provide a single answer as to when and where biomass residues can be used as resources to lower the climate impacts of products and processes. Rather, it is possible to be aware of the methods chosen to study their utilisation, both in terms of when and why an approach may be suitable or not, and in terms of the potential implications such a choice may have on the results and conclusions. All in all, a better understanding of the methods of an assessment allows for a more accurate interpretation of its results. The objective of this thesis could thus be seen as contributing to a use of LCA which can both lead to a better understanding of how biomass residues' management can contribute towards different societal goals, and, based on that, which provides relevant and useful information to decision-making processes.

With that said, there are limitations to the present research. Most notably, this research focuses on climate impacts motivated by the extent of the challenge in keeping global warming levels below 2°C, and aiming at levels below 1.5°C according to the Paris Agreement. The present research thus takes a normative stance in this sense. The threats of climate change are largely the reason why interest in bio-based products to replace fossil-based ones has grown to the extent that it has, and why efforts towards bio-based production have been promoted in different policy initiatives. If bio-based products do not deliver in terms of (low) climate impact compared to fossil resources, one can question their continued promotion.

However, it is not possible to talk about sustainable uses of biomass and biological systems without mentioning other issues, not least biodiversity, but also land and water use, and others. Parts of the research presented here is relevant to types of environmental impacts other than climate change, but in general, the focus is on climate impacts and the goal of mitigating climate change. The climate impacts from the individual case studies cannot be generalised, neither to other types of environmental impacts, or to other types of biomass residues. Instead, general arguments are made for how the life-cycle perspective and LCA can be applied to biomass residues as resources.

Here, I must also acknowledge a certain bias as I have mainly focused on materials and settings relevant to Western Europe and even the Nordic countries. The framing of this research may therefore seem unmotivated from certain perspectives and cases where residual biomass management better fits the picture of a waste problem to be handled. Nevertheless, the ways in which residual biomass valorisation is envisioned to contribute towards more sustainable resource systems, both in policy and in research, warrants consideration.

## 1.3 Research journey through the papers

When I first started this PhD project, I set out to assess the potential environmental benefits of using residual biomass as resources for products and fuels. The general idea was to increase resource use efficiency by making the most out of available resources, and thereby decrease resource extraction and the environmental consequences of it. The idea of using waste and residues seemed to be communicated as a safe option in the biomass and bioeconomy arena which was otherwise muddled by different issues including competition over land, land use change, questioning of climate neutrality, and others. Within this context, I worked with colleagues at the chemistry department on Paper I. Since the use of cellulase enzymes had shown to be a hotspot in lignocellulosic ethanol production, we studied an alternative solution for the enzymes, but also found that the inclusion of bio-based carbon and effects from harvesting of forest residues could greatly impact the results.

As I continued to engage with the literature, cases and research projects, the benefits of residues as resources seemed less self-evident, and different from one case to another. Naturally, there can only be limited improvements from making use of organic residues if we have issues with the environmental impacts of the systems of primary production from which they originate. Somewhere in between the extremes of residual biomass being the solution to bio-based products and fuels, and it being just as problematic as its primary production of origin, my interest grew towards how residual biomass, and products thereof, can be assessed in terms of environmental sustainability. This led to the formulation of Paper II, where I dived deeper into the LCA method and how the inclusion of upstream processes and their environmental impacts can be considered.

It was also evident that there were shortcomings in the traditional use of LCA for assessing biological systems, as had also been indicated in Paper I. I wanted to understand how the method could be adapted to consider the cyclical nature of carbon in biomass and explore how different aspects of uncertainty and preference, both in LCA and in climate impact assessment methods, can affect results and be illustrated. This work resulted in Paper III.

By now it was obvious that the idea of residues as resources was more complex than first thought, but I wanted to come back to the general idea and the principle of closing resource loops. In Paper IV, I therefore departed from the literature on circular bioeconomy and set out to better understand the role of residual biomass in this context, and how the principle and measurement of circularity relate to climate impact studied with LCA. Each of the appended papers thus provides different insights, which form the basis of the following chapters of this thesis.

## 1.4 Outline of this thesis

This thesis continues with chapter two, where important concepts and definitions are introduced, including a definition of biomass residues in general and as studied in this thesis and its appended papers. It then introduces the concepts of circular bioeconomies, and the life-cycle perspective and LCA. Chapter two thus deals with the starting points of the thesis.

Chapters three, four and five each follow a theme related to a research question under which the methods and findings of this thesis are presented and discussed.

Chapter three departs from two perspectives on the function of biomass residues – as feedstocks for valorisation, and as residual materials to be managed – and introduces findings from Papers I and III. In this context, the concept of biogenic carbon and methods for its consideration in LCA are also introduced. Finally, the findings are discussed from the point of view of uncertainty.

Chapter four dives deeper into assumptions of residues as resources free from environmental burden. So called zero-burden assumptions are explained and alternative approaches to consider the full life cycle of biomass residues based on a systems' perspective are presented, based on Paper II, and with examples from Paper IV. I also discuss the relevant method choices based on a general understanding of values in LCA.

Chapter five considers the assessment of residues valorisation towards goals of climate-change mitigation and circular resource systems. This includes methods for climate impact assessment and circularity assessment, and findings from Papers III and IV. I hinge a discussion of the methods and findings on an understanding of models that are imperfect, but that may simultaneously be useful.

Lastly, a concluding discussion is presented in chapter six. Here, I base a general discussion of my findings on the research questions and point to potential future research avenues.

# 2 Concepts and definitions

This thesis departs from a life-cycle perspective on biomass residues and situates itself in the context of increasing interest in circular bioeconomies as a means to rearrange resource systems and mitigate climate change. This chapter therefore introduces and defines the important concepts of biomass residues, circular bioeconomies, and the life-cycle perspective and LCA. The basic principles and constituent parts of LCA are presented based on standards published by the International Organization for Standardization, ISO (2006a, b), but these leave room for interpretation and further development. This chapter gives a very brief introduction to the areas where this thesis aims to better understand and illustrate how different assessment methods and approaches affect conclusions regarding valorisation of biomass residues. Each area, including the relevant approaches and methods used in this thesis and its appended papers, is then further explored and explained in chapters three to five.

## 2.1 Biomass residues

Talking about waste as a resource is not without linguistic complexity, as the very definition of waste often implies that it is unwanted or without value. In this context, the defining characteristics of the waste resources that I call residues is that they have not been the primary intention of industrial production. I furthermore choose to talk mainly about residues as resources, as their use as resources implies that the material is not without function or value, but I make no specific distinction between residues and by-products. The Oxford English Dictionary similarly defines a by-product as "*a substance of more or less value obtained in the course of a specific process, though not its primary object*" (2022a), and residue as "*the remainder, the rest; that which is left*" (2022b), and Merriam Webster defines residue as "*something that remains after a part is taken, separated, or designated or after the completion of a process*" (n.d.-a). Different definitions of waste more clearly imply its lack of value: "*unserviceable material remaining*", "*useless by-products*", "*useless or unsaleable*" (Oxford English Dictionary, 2022c), "*unwanted by-product*" (Merriam-Webster, n.d.-b), "*any substance or object which the holder discards or intends or is required to discard*" (Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives). Within the LCA literature, waste has similarly been defined as material with no or negative economic value (Guinée et al., 2004, Weidema, 2001), or material that does not displace another product (Weidema, 2001), but the term residue has also been used in a similar manner (e.g. Klein et al., 2022).

While the definition of residues is important, it does not necessarily mean that the classification of materials as residues is straightforward. In fact, what materials are considered waste, residues and co-products can vary in time and space:

> "As wastes and residues are not defined by physical properties or chemical composition but rather by process economics, trade and markets, or even perception, what constitutes residual biomass changes over time and across space – a product can be both a co-product and a waste in different places, and it can go from being residual biomass to being one of several desirable, marketable co-products." (Paper II)

The biomass residues considered in this thesis include agricultural and forestry residues such as wheat straw and spruce logging residues from final fellings, residues related to food and forestry industries, including brewers' spent grain and bakery by-products, whey from dairy processing, slaughterhouse and fishery residues, and sawdust and shavings from sawmills. This is not to say that these materials share physical properties nor that they are inherently undesirable as the status as a residual material can be most context dependent. Vural Gursel et al. (2022) classify residues in three categories according to the Netherlands Enterprise Agency (2019): primary or field residues include parts of plants that are left after harvest; secondary or industrial residues are by-products of industrial processing of biomass; and tertiary or post-consumer residues and waste include biological materials that have been used by consumers, including used cooking oil and wastewater sludge. The residues considered in this thesis are thus field and industrial residues according to this classification.

## 2.2 Circular bioeconomies

Residual biomass valorisation is a common denominator in ideas of circular bioeconomies (Stegmann et al., 2020), but there is more to this concept. It can be seen as a combination of circular economy and bioeconomy where the defining characteristics include the use of biological resources rather than fossil resources, the maximising of the value of biological materials by cascading and recirculation (Kardung et al., 2021, European Environment Agency EEA, 2018), and the restoration and regeneration of natural resources (Muscat et al., 2021, European Environment Agency

EEA, 2018). The concept of circular bioeconomy and its promotion in EU policy is not without contestation (Starke et al., 2022), which can be better understood by looking at its origins: circular economy and bioeconomy. The concept of bioeconomy is centred around the reliance on bio-based resources rather than fossil resources, but different and conflicting visions for its aims and means have been identified (McCormick and Kautto, 2013, Bugge et al., 2016). Critique has been raised against a dominant focus on technology at the expense of ecology (McCormick and Kautto, 2013), and on incremental change at the expense of transformative change (Giuntoli et al., 2023, Giurca and Befort, 2023). The concept of circular economy is more ambiguously defined and has been criticised for being broad and vague (Kirchherr et al., 2017, Korhonen et al., 2018) to the extent that there is little of substance to criticise (Lazarevic and Valve, 2017). There are many definitions of it, including the "4R framework": reduce, reuse, recycle, recover, and elaborations of it (Kirchherr et al., 2017), closed material cycles achieved by "*slowing, closing and narrowing resource loops*" (Bocken et al., 2016), minimising the use of resources and energy use and waste creation, and the importance of practices such as design for longevity, maintenance, and repair (Geissdoerfer et al., 2017).

Despite the apparent ambiguity of circular bioeconomies, the concept is used to shape policy initiatives. Many European countries have national bioeconomy strategies in place or in development, as is the case for Sweden (European Commission, 2023), and several bioeconomy initiatives have incorporated circular economy principles (Hadley Kershaw et al., 2021). One example is the updated bioeconomy plan of the EU, which combines the bioeconomy with the circular economy perspective and states that a "*sustainable bioeconomy is the renewable segment of the circular economy*" (European Commission, 2018).

This attention to circular bioeconomies in both policy and research contexts has made relevant a discussion on monitoring their development and performance. As stated in the introduction, aspects of biomass circularity that are pronounced in the literature on circular bioeconomies include sustainable management and regeneration of primary production systems, efficient use of biomass resources by cascading use and recycling, and closing of nutrient cycles by maintaining biodegradability and avoiding contamination by hazardous substances (Vural Gursel et al., 2022, Navare et al., 2021). Various methods have been suggested to measure the circularity of products, but as stated in the introduction, few have been adapted to the characteristics of biological materials and cycles. Among the most commonly mentioned methods for product circularity assessment is the *material circularity indicator*, MCI (Ellen MacArthur Foundation and ANSYS Granta, 2019). It is meant to measure the throughput of material in a product system, i.e. the use of virgin resources, and the lack of reuse, recycling or composting at the end of a product's use phase – all characteristics of linear as opposed to circular resource systems. Other initiatives for monitoring and measuring bio-based circularity focus on different, and often limited, aspects of circular bioeconomies (Navare et al., 2021).

I view the concept of circular bioeconomy and its exact definition both as central and as partly superfluous to the findings presented in this thesis. It is central in framing the relevance of the study of biomass residues as resources to substitute fossil-derived products to mitigate climate change. The point of departure for this work is the current focus on this idea and potential solution in both research and policy, and it is therefore interesting and necessary to better understand the premises for its realisation. The assessment of circularity as presented in chapter five and Paper IV is closely connected to the concept of circularity and its interpretation for biological resources and materials. The valorisation of residual biomass and the climate-impact results presented in this thesis are, however, applicable, and equally valid regardless of the overarching circular bioeconomy concept.

## 2.3 Life-cycle perspective and assessment

The life-cycle perspective and LCA are central to this thesis. This chapter therefore introduces their history and theoretical context, the basics of the LCA method and important concepts, as well as common issues that are relevant to this thesis. In this context, I use life-cycle *perspective* to denote a general idea of how to organise and analyse physical flows related to production systems, and life cycle *assessment* to talk about the method, e.g. as described in the ISO standards. This chapter also introduces the present approach to consequences in LCA, and the calculation approach for biofuels' climate impact that is used in current EU policy, and which is based on the life-cycle perspective.

LCA is the most commonly used method for quantifying the environmental impact resulting from a product or activity. It quantifies the resource use and substance emissions that result from products or activities aimed at fulfilling a certain *function*, and relies on available knowledge from other research fields to say something about the resulting environmental impacts. It can thus be useful in trying to better understand the environmental impacts related to different options available to us, whether it has to do with choosing how to eat, transport ourselves, or in deciding on the most effective measures to improve a process, product or service from an environmental perspective. LCA is thus clearly connected to decisions it can support by illustrating the environmental impacts related to different alternatives. The field dates back to the

1960s and 70s, when early assessments focused on energy and resource use, and waste generation, but the term life cycle assessment was not set until 1990 (Bjørn et al., 2018). The first global ISO standards for LCA were released in 1997 and 1998, and later updated in 2006 to present a general framework for LCA.

LCA has connections to other research fields and types of methods. The life-cycle perspective is a systems perspective, and LCA a systems analysis tool. A *system* can be understood as a set of interconnected elements that are organised around a function or purpose (Meadows, 2009). In systems' analysis, important aspects of study are how the different parts of a system interact, and how the system interacts with other systems (Ghosh, 2015). In LCA, this includes understanding how a change in one part of the system affects the whole to avoid unintended outcomes and suboptimization, and how the life-cycle system interacts with its surroundings, and other systems, by the use of resources and the emission of different substances. The life-cycle perspective and LCA also have links to the field of industrial ecology, which is aimed at the study of industrial processes from a systems' perspective, drawing on biological systems as models for industrial processes, focusing on flows of resources and energy, and driven by the ambition to support sustainable modes of organising societies (Graedel and Allenby, 1995). It shares with the life-cycle perspective the view of a system from 'cradle to grave', and the encompassing of material and energy flows (Lifset, 2006). LCA can thus be seen as one approach and method to perform studies within the field of industrial ecology, and to advance its goals.

LCA is also a form of *technology assessment* that involves both the production of knowledge and the valuation of that knowledge, and recommendations based upon it to inform decision-making processes (Grunwald, 2009). More specifically, LCA is an *environmental systems analysis tool* that differs from other tools in its focus on the environmental impacts of a product or function, and its life cycle (Finnveden and Moberg, 2005). Neither technology assessments nor environmental systems analysis tools can provide answers to the problems to which they are applied. This type of method or approach can provide "*knowledge, orientation, or procedures on how to cope with certain problems at the interface between technology and society but it is neither able nor legitimized to solve these problems*" (Grunwald, 2009). This is due to both theoretical and practical obstacles (Finnveden, 2000) that will be discussed throughout this thesis.

Unlike other environmental systems analysis tools, LCA is centred around the concept of a functional unit, which describes a sought function or service. The function defines the life-cycle system, its system boundaries, and the relevant flows of mass and energy. The environmental impacts of the system are quantified by adding an environmental impact assessment of the resources and emissions that cross the system boundaries. In LCA terms, the process of combining the emissions of different substances to a single score for a certain environmental impact category is called characterisation. Here, an LCA relies on knowledge from different fields for creating characterisation methods, and when there are several options available, it is up to the practitioner to choose and explain their choice. There may be several characterisation methods available, and they may quantify different types of impact at different points of the cause-effect chain of an environmental mechanism (ISO, 2006b). For instance, climate change can be quantified as an increase in radiative forcing, which is a measure of the change in the Earth's energy balance resulting from e.g. an emission of a greenhouse gas (Myhre et al., 2013a). Such a method is considered a midpoint indicator. As an alternative, the damage of climate change to human health and ecosystems may be quantified with an endpoint indicator that is at the end of the cause-effect chain (Rosenbaum et al., 2018b).

Overall, the presence and behaviour of the LCA practitioner is crucial to the assessment. This is indicated by the importance of the interpretation phase of LCA according to ISO standards, which connects to the other phases of goal and scope definition, life-cycle inventory and environmental impact assessment (ISO, 2006a). It is also important since the practice of applying LCA to provide information for decisions according to ISO standards leaves room for interpretation.

### 2.3.1 Method considerations

One of the most commonly discussed issues within LCA arises when flows and processes are shared between several products and life cycles, often referred to as the multi-functionality issue (Pelletier et al., 2015, Ekvall and Finnveden, 2001). This issue occurs more often than not, and it is especially relevant in the context of this thesis since residues per definition share processes with other valuable outputs. In addition, different choices of the method used to handle the multi-functionality issue can lead to diverging LCA results (Cherubini et al., 2011b). There are generally two strategies at hand to deal with multi-functionality: either the system boundaries are expanded to incorporate all the relevant products and functions, which means that the functional unit is expanded in the same way, or a rule is applied to artificially create a delineated system with only the sought product as output. In LCA terms, the former is system expansion, and the latter is made possible by allocation (ISO, 2006a).

The ISO standards give a prioritisation for how to go about the multi-functionality issue, which is both widely adopted and criticised (Schaubroeck et al., 2022, Schrijvers et al., 2016), and with some dispute over its rationale and interpretation (Pelletier et al., 2015). In general, system expansion is to be prioritised, but it cannot be applied if LCA results are sought for an individual product or function. In the context of multifunctional production systems such as biorefineries, the choice of a functional unit that best represents the different functions delivered with the product portfolio often stands in contrast to a functional unit which is comparable to other means of production, or other biorefineries (Ahlgren et al., 2015). There is, however, the choice of substitution where a by-product is assumed to replace an alternative product with the same function, and the environmental impacts related to that alternative product is subtracted from the main system along with the by-product function. The remaining function, and environmental impact, is thus that of the main product. The term substitution is sometimes used interchangeably with system expansion, and while they are not the same thing, substitution is generally regarded as a type of system expansion (Vadenbo et al., 2017). Next in the ISO order of priority is allocation; first by following "underlying physical relationships" (ISO, 2006b), and secondly by other means such as economic relations. This approach partitions different shares of the total environmental impacts to different output streams, and thus also allows for creating results for a single product or function. These concepts are all relevant to the application and scrutiny of LCA in the appended papers, as will be explored further in chapters three and four.

Another issue within LCA that is relevant to the studied cases in this thesis is that of the temporal dimension of the life cycles of bio-based products. For instance, the possibility to let biomass residues decay in fields and forests is difficult to consider at a single point in time, since processes of decay tend to happen over time. The life-cycle inventories which form the basis of the quantitative assessment are, however, often created as a single snapshot that does not account for resource uses and emissions happening at different points in time. Similarly, the environmental impacts calculated by the combination of the life-cycle inventory and the chosen characterisation methods are commonly displayed as a single value, potentially with an indication of uncertainty. While the ISO standards provide no guidance in this matter, methods of *dynamic* LCA, which aim to account for the temporal dimension, have been developed (Levasseur et al., 2010). A deeper investigation into the temporal aspect of the life-cycle perspective for biomass residues is made in chapters three and five.

### 2.3.2 Attributional and consequential approaches

In this context, it may also be relevant to briefly explain how consequences are considered in LCA in this thesis. When LCAs are concerned with consequences of actions they can be referred to as consequential LCA (Finnveden et al., 2009, Ekvall et al., 2005). The focus of such studies concerns the consequences of an action or decision, all else being equal. This is opposed to attributional LCA, which instead is concerned with a snapshot of the world, and with attributing a certain share of physical flows to a function or product system. It has therefore been argued that consequential LCA is more relevant to policy (Plevin et al., 2014b), since policies aim to have consequences. It has also been argued that the approach used should be adapted to the question asked, and that there are different settings where both attributional and consequential approaches are valuable (Ekvall et al., 2005, Brander et al., 2019, Yang, 2016, Brandão et al., 2014).

I have chosen not to label my studies as attributional or consequential LCA studies in line with the ISO standard for LCA, but instead focus on explaining the rationale for setting up each study. In Papers I and IV, the approach is attributional in the sense that I focus on a snapshot of all resource use and substance emissions related to the activities of a product's life cycle. This approach allows for a comparison of the results to other attributional assessments, including reference values for fossil fuels. Paper II does not contain an LCA, and as such cannot be labelled with either an attributional or consequential approach. It does, however, depart from the need to handle multi-functionality in an attributional sense. In Paper III, the functionally equivalent scenarios that are compared could be seen as constructed based on the consequences of i) harvesting or ii) not harvesting forest and agricultural biomass residues. The assessment of the resulting climate impact of those consequences, however, is not integrated into a single result, but as a comparison between two scenarios that are each studied with an attributional approach.

### 2.3.3 Approach of the Renewable Energy Directive, EU RED

In contrast to LCA, as defined in the ISO standards and the method discussions and developments beyond them, a different track can be seen in more streamlined applications of LCA. A highly policy-relevant example in this context is that of the greenhouse gas calculation method for liquid biofuels in the EU renewable energy directive, RED (Directive (EU) 2018/2001 of the European Parliament and of the Council of 11 December 2018 on the promotion of the use of energy from renewable sources (recast)). In general, the RED promotes renewable energy in the member states as a strategy to meet the climate targets of the EU and of the Paris Agreement. As part of the directive, the use of residual biomass for biofuels for transportation is promoted by gradually increasing targets for the share of such biofuels that are produced from residual feedstock (article 25.1). The directive allows for certain residual feedstocks to be counted as double towards these targets, including straw, tree tops and branches, and different industrial by-products which are not fit for food or feed applications (Annex IX)[2].

In addition to these targets, the RED defines increasingly challenging targets for the level of climate impacts from biofuels. This is where the directive defines a method for calculating the relevant greenhouse gas emissions and climate impacts. This method is based on a life-cycle perspective, but differs from ISO-compliant LCA in that it specifies, for instance, how to deal with multi-functionality, where to draw system boundaries, what characterisation method to use, and the fossil fuel comparator. The RED sets restrictions concerning the type of land that residual biomass can be harvested from which are aimed at avoiding unwanted negative effects in terms of biodiversity loss, soil quality loss, and increased greenhouse gas emissions. If such sustainability criteria are met, however, the calculation method of the RED states that when biomass residues are considered as feedstocks, processes prior to the collection of residues are to be disregarded (Annex V). The RED method and the rationale and importance of this type of method choice are further explored and discussed in chapters four and five of this thesis.

[2] Amendments of the directive are being discussed, and its rules may therefore change.

# 3 Alternative routes for residues

The use of biomass residues as feedstocks has been identified as one of the most promising options for achieving bio-based energy carriers with low climate impacts (Creutzig et al., 2015). Biomass residues can be cheaper than primary biomass, but they can also be more dispersed and heterogenous which may lead to additional demands for resources and energy to convert the material to products. The valorisation of residual biomass can be implemented in different ways and employing different technologies, and some residues can also be left to decay in forestry and agricultural environments, and thereby contribute functions other than valorisation of the same material. This chapter introduces these different perspectives on residual biomass use and how they can be studied in LCA. It also presents findings for the case of lignocellulosic ethanol production from Papers I and III, including the impact on results and conclusions from the choice of how to consider the climate impacts from the greenhouse gas emissions that originate from biomass. Finally, the concept of uncertainty in LCA is used as the basis for a discussion of the findings of this thesis and the data used to produce them.

## 3.1 Functions of residues as resources

To define a suitable functional unit for biomass residues as resources, it is useful to depart from possible goals of the study. Oldfield et al. (2018) argue that a functional unit based on the product function is appropriate for assessing residues' valorisation in a circular economy. Then the sought function is the focus of the assessment, and different potential feedstocks and processes can be compared as illustrated in Figure 1B. Contrastingly, Hanssen and Huijbregts (2019) argue that the relevant question for residues concerns how the materials are best managed. The functional unit is then instead defined as management of a certain material, and different management options can be compared as in Figure 1A. This latter perspective is more in line with classical waste-management LCAs. While these approaches and perspectives are different, they are both relevant. Oldfield et al. (2018) also show how conclusions can vary with the chosen functional unit, which is a general argument for keeping both perspectives in mind (Escobar and Laibach, 2021). The definition of the functional unit can – and should – thus depend on the question asked.

This chapter illustrates both these perspectives for residual biomass based on the case of lignocellulosic ethanol production from logging residues and straw. In chapter 3.2, the point of departure is the possibility to valorise residual biomass into products, and the functional unit corresponds to product function (Figure 1B). This way, the climate impact of residue-based products can be assessed and compared to that of other products which provide a similar function, such as fossil-based products, or as in the studied cases in Paper I, the same product produced in a different process. The focus is thus on identifying potential climate hotspots and on bio-based products' comparison to fossil-based ones. The perspective applied in 3.2 and in Figure 1B does not, however, aim to answer questions regarding how the residual material is best managed. This perspective is instead the basis for chapter 3.3. Here the focus is on comparing residues' valorisation to an alternative scenario which, for the logging residues and wheat straw studied in Paper III, does not involve their harvest, but instead includes their decay at the final felling and harvest sites.

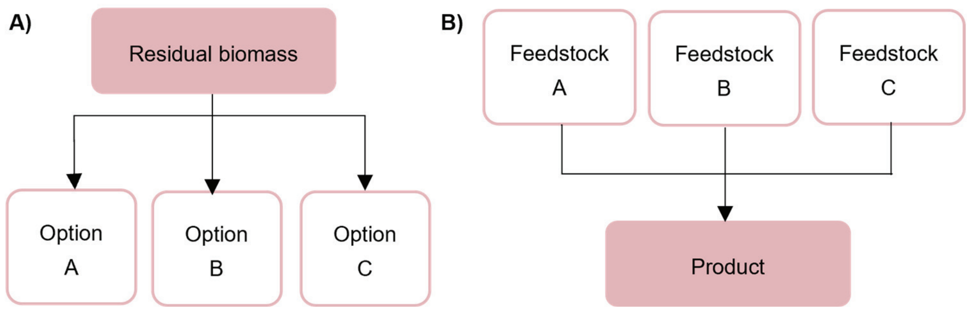


**Figure 1**
Different perspectives on biomass residues as resources. A: residual biomass as a material to be managed. B: residual biomass as a resource for valorisation, and as a feedstock comparable to other feedstocks.

## 3.2 Assessing valorisation options

Many biomass residues from forestry and agriculture, such as branches, bark, straw and husks, are materials rich in cellulose, hemi-cellulose and lignin. These are sometimes referred to as *lignocellulosic* materials. Unlike the feedstocks used for first-generation biofuels that are mainly based on starch and sugars from food crops, the processing of lignocellulosic materials into fuels and other products typically requires pretreatment to make the carbohydrates available to enzymes, followed by an enzymatic hydrolysis or oxidation for their conversion to sugars (Carlqvist, 2022). The use of enzymes to hydrolyse cellulose has been identified as a hotspot in lignocellulosic ethanol production (Slade et al., 2009, MacLean and Spatari, 2009). In Paper I, we therefore compared two production alternatives for lignocellulosic ethanol made from logging residues: one where the enzymes are purchased from an external producer, and one where the enzymes are produced by fungi in an integrated enzyme and ethanol production process. The integrated process includes the use of logging residues as feedstock not only for ethanol production, but also for enzyme production. In the integrated process, the whole fermentation broth is used, removing the need for separating and stabilising the enzymes.

The findings in Paper I align with previous research in terms of the external production of enzymes as a hotspot for greenhouse gas emissions, but also indicate the potential to lower them with an integrated production process. The comparison between the production alternatives, however, depends mainly on the data assumed for the external enzyme production. For instance, the choice of data source for the enzymes and different assumptions regarding their dosage and the energy carrier used in their production affects the comparison and the conclusions. More recent research and development of the enzyme products available has similarly shown that the climate impact of this specific input can be lower than previously estimated. New enzyme products are produced with lower climate impacts, and though their formulation requires a higher dosage of the enzyme cocktail, the significance of the enzyme production relative to the climate impacts of the total lignocellulosic ethanol production is lowered (Karlsson, 2018, Karlsson et al., 2017, Gilpin and Andrae, 2017). With a lower impact associated with the enzymes, the collection and transportation of logging residues constitute a large share of the climate impact of the ethanol. This source of climate impact could potentially be lowered using fuels with lower climate impacts, and could constitute an important point for improvement in the integrated production case. Overall, the difficulty in identifying reliable data for external enzyme production, a technology under development, was an obstacle in assessing the potential benefits of an integrated production set-up in Paper I. I continue this discussion of uncertainty in chapter 3.5. The results, however, also indicated that the cases of integrated ethanol and enzymes production could lead to costs of ethanol that are only slightly higher than, or comparable to, those of production with external enzymes.

In line with the valorisation perspective, the functional unit in Paper I was based on the main product, ethanol. The by-products, electricity and lignin pellets, were dealt with both by substitution and by allocation based on energy content. The chosen approach to deal with multi-functionality affected the absolute results for ethanol, but the comparison between the external and integrated production processes was not affected, and neither was the comparison to a fossil fuel reference. The focus on ethanol as the main product motivated the method choices in Paper I, but it would also have been possible to avoid the multi-functionality issue by including all the relevant functions in the functional unit. This was the approach in Paper III, where the same type of integrated enzyme and ethanol production process for logging residues was compared to the use of fossil fuels in a reference scenario. The first results from Paper III confirm that the climate impacts related to the scenario where logging residues are used as feedstock for ethanol production are significantly lower than those of a corresponding reference scenario with fossil fuels. Several sensitive method choices and parameters were, however, identified in Paper III, which will be further discussed in the remainder of this chapter, and in chapter five.

## 3.3 Assessing leave-be and management options

Valorisation of biomass residues require their harvest or collection, and harvesting of biomass residues has generally been coupled with a certain measure of precaution. Biomass contains both organic matter and the nutrients needed for the organism to grow, and therefore harvest is related to the removal of these components from the place where the biomass grew. Leaving biomass residues (primary or field residues) behind at harvest or final felling is one potential way of minimising the loss of valuable nutrients and preventing loss of soil quality (Andrade Díaz et al., 2023, Cherubini et al., 2009, Ranius et al., 2018). This principle is also present in ideas of future circular bioeconomies where there appears to be dual expectations on biomass residues to both act as that bridge of nutrient and material circulation to soils, and to provide an extended feedstock base for valorisation into products (Bos and Broeze, 2020, European Environment Agency EEA, 2018). In the literature analysed for Paper IV, it is possible to see that long-term productivity and health of primary production systems is considered essential for the long-term viability of circular bioeconomies. Biomass residue management could contribute towards such a goal both by avoiding harvest of residues and instead leaving them behind, and by returning process residues of different kinds, such as wood ash from incineration or the digestate from anaerobic digestion, to soils. For this to be possible, the biological materials cannot be contaminated with substances or materials hazardous or harmful to the environments to which they are to be returned.

Another argument for considering leaving biomass residues behind can be made from a climate-impact perspective, especially for long-rotation biomass such as trees. In this case, the alternative of not harvesting the residues effectively binds the carbon they contain for a longer period of time than if they were to be used for energy or other purposes where they would soon be combusted. These aspects are considered in the design of the scenarios in Paper III by the explicit comparison of either harvesting and using biomass residues as feedstocks, or by leaving them to degrade in a reference scenario. This approach thus covers the perspective illustrated in Figure 1A: each functionally equivalent scenario considers the same amount of biomass residues – logging residues or straw – and therefore concerns the question of how different management options for the residues compare in terms of, in this case, climate impacts. The scenarios are, however, also made comparable in the sense that they deliver the same functions, they are *functionally equivalent*. In the biofuel scenario, the residues are harvested and used as feedstock for the ethanol production system. In the reference scenario, they are instead left to decay where they are, and the same functions as in the biofuel scenario are provided by means of other products based on fossil resources. Including such a reference scenario for biomass use is generally advised (Koponen et al., 2018) to enable comparisons of different management options. In Paper I, the reference scenario for the residual biomass use is not explicitly illustrated, but included in a sensitivity analysis by considering the additional climate impact resulting from the valorisation of the residues as an immediate emission of carbon to the atmosphere that would not have happened if the residues had not been harvested (see 3.4 for more details).

In terms of reference scenarios, there could also be other relevant pathways to compare valorisation of residues to. For instance, (Pfau, 2015) suggests the term *resource use change* to denote the redirection of residual biomass from one use to another. Leaving residues to degrade in natural or semi-natural environments is not always a realistic option for industrial or post-consumer residues. Instead, their present or alternative use could denote the reference scenario. The focus in resource use change lies on the present use of biomass residues from which the residues will disappear as a consequence of the new application. Other terms used to describe the same principle include counterfactual[3] (Welfle et al., 2017), reference (Koponen et al., 2018), or alternative utilisation (Tonini et al., 2016). In principle, the valorisation of a residual biomass towards a certain use can be compared both to a scenario where residues are not harvested or collected, and to other valorisation options.

[3] To make matters more confusing, the term counterfactual is also used to describe a product that can be substituted. (Hanssen and Huijbregts, 2019)

## 3.4 Climate impacts of biogenic carbon

The inclusion of a reference scenario to biomass utilisation can thus be important, for instance in order to consider the different impacts on climate from carbon from biomass residing in the biosphere or the atmosphere. This connects to a methodological debate in the field of LCA, and greenhouse gas accounting more widely, concerning the climate impacts of biogenic carbon. A common assumption is that of the climate neutrality of biogenic carbon derived from the cyclical nature of the life cycle of plants and biomass (Lamers and Junginger, 2013). The plant grows and sequesters carbon as $CO_2$ from the atmosphere, and then dies to decay or by other means have its carbon content released to the atmosphere. The sequestration and emission of biomass carbon could therefore be viewed as a net zero exchange with the atmosphere over a certain time period. The climate neutrality assumption has, however, been challenged for different reasons, not least when applied to long rotation biomass such as trees. The argument that uptake and emissions are equal over time appears less relevant when the lifetime of the biomass and the residence time of the biogenic $CO_2$ in the atmosphere is longer. This is because the climate impact due to carbon temporarily in the atmosphere, or the climate impact avoided by carbon temporarily stored in biomass, cannot as easily be disregarded when the temporary state is stretched out in time. A time perspective which does not cover many subsequent biomass life cycles can instead lead to a chicken-and-egg dilemma (Albers et al., 2020): what came first, the growth and sequestration of $CO_2$, or the harvest and emission of $CO_2$?

The study design in Paper III avoids this issue by assuming that final felling of spruce happens equally in the biofuel and reference scenarios, and new spruce trees are assumed to grow equally in the two scenarios. Because the study is concerned with the difference between the two scenarios, the aspects that are identical between them are not relevant to the conclusions regarding their comparison. The difference between the scenarios is instead that the logging residues are harvested and used as feedstock for ethanol production in the biofuel scenario and left to slowly decay in the reference scenario (compare to the perspective illustrated in Figure 1A). To reflect this difference in the climate impact assessment, all biogenic carbon emissions are considered in Paper III. To be more precise, parallel calculations are made where biogenic carbon is assumed to be climate neutral, and where it is not. The same set-up is applied to the case of wheat straw.

There are, however, different ways of including biogenic carbon in greenhouse gas balances and life-cycle inventories. The approaches used in Paper III include the use of $GWP_{bio}$ factors (Cherubini et al., 2011a, Guest et al., 2013) and the modelling of biogenic $CO_2$ emissions and uptakes over 100 years' time. The $GWP_{bio}$ factors are created to be applied to biogenic $CO_2$ emissions just as other GWP factors. They essentially consider the harvest of biomass to precede regrowth of the same type of biomass, and measure the climate impact related to a temporary stay of biogenic $CO_2$ in the atmosphere (Cherubini et al., 2011a). Similarly, the factors by Lindholm et al. (2011) for logging residues that are applied in Paper I include the climate impact caused by the temporary stay of biogenic carbon in the atmosphere compared to a scenario where residues are left to decay in the forest. This scenario comparison is thus implicit in Paper I in the calculations where these factors are applied (called *SOC factors*[4] by Lindholm et al. and in Paper I), and explicit in Paper III. The factors used in Paper I are specific to logging residues, and the $GWP_{bio}$ factors used in Paper III are general for biomass with a certain rotation time. Both imply the compression of greenhouse gas emissions and their climate impacts over time to a single number.

The calculations based on explicit modelling of biogenic carbon in Paper III, however, also entail life-cycle inventories with a time resolution. This means that the decay of the residual biomass that is left behind is considered on a yearly basis, and therefore the reference scenario includes greenhouse gas emissions for several years. The resulting climate impact can then be calculated as a GWP result to be compared to any other result for the same functional unit. This consideration of a temporal resolution in LCA has created a field of dynamic LCA (Beloin-Saint-Pierre et al., 2020, Levasseur et al., 2010, Sohn et al., 2020) and it has been argued as critical to LCAs guiding decisions (Lueddeckens et al., 2020). Here the focus is on the implications of including biogenic carbon and residues' decay, but a continued discussion of other implications related to the temporal dimensions can be found in chapter 5.

The inclusion of biogenic carbon effects for logging residues by static factors in Paper I affects several conclusions for the lignocellulosic ethanol production studied. To properly interpret the results, it is important to highlight again that the factors applied in Paper I consider the contribution of logging residues to carbon stored in soil, as well as the delay of greenhouse gas emissions from the temporary storage of carbon in logging residues in the forest before they decay. The climate impact of the ethanol from the integrated production is either lower than or approximately in line with the highest climate impact observed for ethanol using purchased enzymes in our study, and all depending on the assumed effect by harvest on soil organic carbon and the climate impact of biogenic $CO_2$. This effect varies with the different assumptions made by Lindholm et al. (2011) for condensing long-term greenhouse gas fluxes into a value

[4] Note that in this text, I differentiate between soil organic carbon (SOC) as stored in the soil, below ground, and carbon temporarily stored in decaying biomass or litter, for instance as piles of logging residues in the forest.

without time resolution. Since the integrated ethanol and enzyme production uses part of the forest residue feedstock for the enzyme production, it requires more forest residue per unit of produced ethanol than the process relying on purchased enzymes. Notably, this approach drastically affects the climate impact results for all the alternative ethanol production set-ups studied in Paper I.

The results of Paper III show that the method for including biogenic carbon potentially matters to the conclusions drawn for residues' valorisation. The consideration of biogenic $CO_2$ is important to the conclusions in the case of logging residues, but not in the case of wheat straw. The results indicate that the biofuel scenario for wheat straw results in lower climate impact than the reference scenario. For logging residues, however, both the inclusion of biogenic carbon and the method for its inclusion matter to the comparison between the biofuel and reference scenario. Without biogenic carbon, the biofuel scenario results in lower climate impacts at the 100-year time horizon. The outcome is, however, the opposite using the $GWP_{bio}$ factors where the results indicate approximately 20-35% greater climate impact in the biofuel scenario. When all biogenic carbon fluxes are considered in a time-dependent inventory over 100 years, the comparison is more even. With 100-year timeframe, the scenario with the lowest climate impact depends on the assumed greenhouse gas intensities of the fossil fuels, while the assumed decay rate for the logging residues is of lesser importance. Time-dependent modelling with explicit reference scenarios may seem more demanding in terms of the required expertise, time and data. Based on the results from Papers I and III, however, the case can also be made that it can facilitate interpretation and significantly increase the understanding of important method choices and potential hotspots. These results also show the necessity to consider different perspectives before concluding on valorisation options for residual biomass, including both reference scenarios and alternative valorisation processes.

## 3.5 Data and uncertainty

Many aspects come into play in assessing whether the use of biomass residues as feedstock for ethanol production can be considered a successful climate impact mitigation strategy. So far, this chapter has illustrated how some of these aspects affect the conclusions for biomass residues valorisation for lignocellulosic ethanol production, including the process design for valorisation, the method for including biogenic carbon flows, and the choice of what to compare to. To sum up, the answer is coupled with uncertainty, and the following section is dedicated to understanding better what that implies.

In this context, *uncertainty* is what we do not know. We have uncertainty due to incomplete knowledge of the world, for instance due to the inability of a model to precisely describe processes in the real world. It is also possible to describe the nature of uncertainty as precision or accuracy, where precision describes reproducibility or spread in results, and accuracy describes distance to a target (Rosenbaum et al., 2018a). The concept of uncertainty is sometimes meant as encompassing *variability*, which stems from inherent fluctuations in the real world. Variability can in principle be measured but not reduced, whereas uncertainty can be reduced but not eliminated.

A first framework to classify different types of uncertainty and variability in LCA was presented by Huijbregts (1998). In this framework, uncertainty takes the form of parameter uncertainty, model uncertainty, and uncertainty due to choices. Variability covers spatial variability, temporal variability, and variability between objects. In this context, I treat the different types of variability as part of the different types of uncertainty, in line with Rosenbaum et al. (2018a). For instance, parameter uncertainty encompasses both variability and uncertainty in model input parameters. There may be parameter uncertainty due to incomplete knowledge of a parameter, e.g. due to imprecise or outdated measurements, or even lack of data. Björklund (2002) divides parameter uncertainty into data inaccuracy, data gaps, and unrepresentative data. Model uncertainty, on the other hand, is related to differences between processes in the real world and the life-cycle model of them. For instance, the inability of traditional LCA to consider resource use and emissions over time, e.g. as in the case of biogenic carbon flows, may add to model uncertainty. Lastly, uncertainty due to choices, or scenario uncertainty (Huijbregts et al., 2003), refers to uncertainty stemming from choices based on values. The choice of functional unit or allocation procedure can be mentioned as an example.

In general, the uncertainty of choices such as that of allocation approach are commonly treated as *sensitivity*. The concept of sensitivity is often and in this text understood as "*the effect of a certain change in input on the output applying a predefined variation without considering uncertainty*" (Rosenbaum et al., 2018a). It can be tested by varying an input parameter and observing the outcome of the model in a local sensitivity analysis (Pichery, 2014). Levasseur et al. (2016) similarly suggest communicating on uncertainty and *ambiguity* where the latter indicates how the result depends on choices made by the LCA practitioner.

Different types of uncertainty can thus be understood as stemming from natural fluctuations in the real world, measurements of the real world and unrepresentative data, from the model used to make sense of data and the world, and from choices. The presence of choices and specifically value-choices is an important theme in the next chapter, and I will return to the values that cause uncertainty due to choices in chapter

4.3. Additionally, assessments of future technologies are also coupled with uncertainty about future developments. In such studies, sometimes called prospective LCAs, the way in which a technology will develop and mature, and the way in which surrounding systems will develop, is uncertain (Arvidsson et al., 2018). The relevant sources of uncertainty in the case of lignocellulosic ethanol production can therefore be understood both as parameter, model and choice uncertainty, and as uncertainty about the future.

All these mentioned sources of uncertainty are present throughout this thesis work. The difficulty in choosing data and understanding whether it actually represents what it is intended to, is not least visible in the choice of data for externally produced enzymes in Paper I. Additionally in this case, the sensitivity of that data to different assumptions is also difficult to fully understand, as insight into the production of data is limited. In this context, openness and transparency on the data and methods applied, as well as their interpretation, can contribute to making research more reliable in the sense that others can follow, reproduce, and criticise it (Elliott, 2022). As an attempt at providing more transparency, the data used for the assessments in the appended papers is gathered from scientific literature and publicly available datasets. The numerical inputs are made available along with the published papers, for instance as part of calculations in Microsoft Excel files. The use of such data not only makes it possible for others to find it, but also to give a detailed account of the applied numerical values in the published papers. It is not necessarily so that publicly available LCA data is traceable in terms of underlying data and methods, but within the context of this thesis, it is one attempt at transparency and traceability.

In general, the approach to illustrate sensitivity and discuss uncertainty has guided this thesis work. The identification of the different sensitive parameters and choices that are potentially decisive to comparisons and conclusions can be seen as the main outcomes. Sensitivity analyses based on alternative data sources are part of Papers I and III. In Paper I, alternative data show the potential impact that different enzyme production systems may have on the results. These different data sources also indicate uncertainty about future developments, as they indicate different potential dosages of the enzymes, and the energy carriers used in their production. Similarly, Paper I includes cases with a higher efficiency (activity) of the internally produced enzymes that could serve to illustrate a development and maturing of the technology. The choice of data for enzymes is of course not the only source of uncertainty, and the general approach of sensitivity analyses is present and important in all the LCA studies of this thesis.

In Paper III, a sensitivity analysis is included in the low and high cases that are studied for the biofuel and reference scenarios. This approach illustrates how the comparison between scenarios is affected by assumptions of more greenhouse gas intensive product types and production routes such as those for fossil fuels, electricity, and important process additives. Such uncertainty could also indicate uncertainty about future developments, however, based on currently available production systems. Additionally, the high and low cases include different methods for estimating the decay rate of forest residual biomass but based on the same geographical area. The range between the low and high cases therefore includes an indication of model uncertainty. The decay processes also depend on locally specific conditions that are not fully covered in the decay models. The applied parameters that describe decay rates are therefore coupled with different types of variability and further uncertainty, even though the results are based on a specific geographical location in Sweden. Several factors that affect the decay rate of logging residues and other types of biomass residues, such as temperature and precipitation patterns, could also change as a consequence of future climate change. The ranges presented in Paper III are, therefore, also for this reason, coupled with uncertainty about the future.

The findings presented in this chapter show how a combination of method choices and assumptions can affect what appears to be the better way to make use of residual biomass, and specifically forest residual biomass, from a climate-impact perspective. These aspects range from the design of the valorisation process to method choices for considering the climate impacts of biogenic $CO_2$, and uncertainty about the future is ever present. The question of where the residual biomass comes from, and the potential climate impacts and other sustainability issues of such processes, is mainly outside the scopes of Papers I and III due to their comparative approaches. Instead, the next chapter looks further into this aspect of biomass residues as resources.

# 4 Residual feedstocks and zero-burden assumptions

An important perceived advantage of biomass residues as resources is that issues related to primary biomass production can be avoided, including land use change and carbon debts. In Sweden, the use of HVO[5] as a transportation fuel with low greenhouse gas emissions that can be made from biomass residues has increased substantially in the last decade. The Swedish Energy Agency states that the greenhouse gas emissions related to the HVO used in Sweden are relatively low because the feedstock mostly consists of residues and waste – in 2021, 63% of the HVO was produced from animal fats (Energimyndigheten, 2022). These animal fats include slaughterhouse waste and fish waste, and most of these originate outside of Sweden. The calculation of the greenhouse gas emissions of the HVO used in Sweden follows the assumption in the EU RED that the processes leading to the creation of residues – their *upstream* processes – are not considered. In the context of the RED, this is an expression of a political priority aimed at making use of biomass residues as feedstocks for biofuels rather than intentionally grown biomass for bioenergy purposes. The assumption of residues as disconnected from the impacts associated with their upstream processes is, however, also visible in other contexts. When such assumptions are made in LCA studies, the underlying reasons are not always clear or explicit. This chapter is therefore dedicated to the scrutiny of assumptions of residues as free from the impacts arising from upstream processes, and to alternative ways of considering such processes and impacts. It also presents a discussion of allocation approaches and rationales based on values in LCA, and of the implications of including upstream processes in studies of residual biomass valorisation.

[5] Hydrotreated vegetable oils, though both vegetable and animal oils are used as feedstock.

## 4.1 Zero-burden assumptions

Both current residue-valorisation assessments and traditional LCAs of waste management tend to disregard the upstream processes of residual biomass but based on different rationales. Traditionally, LCAs of waste management options commonly depart from the treatment of one unit of waste and compare different available treatment alternatives for that waste (compare to Figure 1A, departing from a residual material and comparing management options). The processes that lead to the creation of the waste material, and their environmental impacts, are the same regardless of the waste management scenario. The comparison of one management alternative to another is thus not affected by the upstream processes, because they are assumed to be equal regardless of the management alternative. This line of reasoning thus implies that since any process that is identical in scenarios does not affect their comparison, such processes can be left out without compromising the conclusions drawn from a comparison. This way, the processes in which the waste is created can be disregarded in studies of waste management alternatives in a so-called *zero-burden assumption* (Clift et al., 2000, Finnveden, 1999). This is, however, not the same as disregarding the previous parts of the life cycle, the upstream processes, outside of such a comparative assessment. Therefore, if the study does not aim to compare management options for a certain material, this type of zero-burden assumption is not valid.

Instead, Paper II clarifies and strengthens the argument that zero-burden assumptions for residues are to some extent already, and will increasingly become, obsolete. This is because biomass residues are, in many contexts, increasingly considered valuable resources (Oldfield et al., 2018, Pradel et al., 2016, Djuric Ilic et al., 2018, Wiloso, 2015). In Paper II, the different possible grounds for regarding residues as burden-free within the logic of LCA are scrutinised. One potential explanation as to why the logic behind different assumptions that cut off waste and residues from their upstream processes are not always made clear in LCA practice, could be that there are several conditions under which residues can be regarded as burden-free. For instance, the definitions of waste as materials of no value, as discussed in chapter two, could lead to an economic allocation of zero to waste materials. This is in line with the methods available for handling multi-functionality in LCA, but it is not the same type of assumption as traditional zero-burden assumptions for waste management, as explained above. It may be in the grey area where residues – not waste – are used as resources that this assumption is sometimes used in cases where it is not entirely applicable, as residues are not without value.

## 4.2 Allocating impacts to residues

The question then remains how the upstream processes of residual biomass can be considered in assessments of residues' valorisation. As an alternative to the traditional zero-burden assumptions for waste management studies, and to an economic allocation of zero to waste materials, the use of residues as resources can be considered a case of open-loop recycling (Paper II). In the ISO standards (2006b), open-loop recycling is when a material from one product system is recycled in another product system, as illustrated in Figure 2, and the recycled material undergoes a qualitative change in the sense that it cannot be used again within the first product system. With several different product systems, an issue of multi-functionality arises. It is at the delineation between the first product system, and the second – which makes use of the residual material as a resource for a new product or function – that the upstream impacts of residues should be considered. As with any issue of multi-functionality, however, there are different ways to approach it. In Paper II, we reviewed the literature of LCA applied within the industrial sectors of bakery, brewery, dairy, fishery, slaughterhouse, sawmill and pulp mill, to better understand how biomass residues as resources are handled in both quantitative and qualitative terms. More specifically, we looked at the qualities of the products and residues that were used to deal with the multi-functionality issue, the reasons for choosing a certain approach to deal with this multi-functionality, and the impacts allocated to residues.

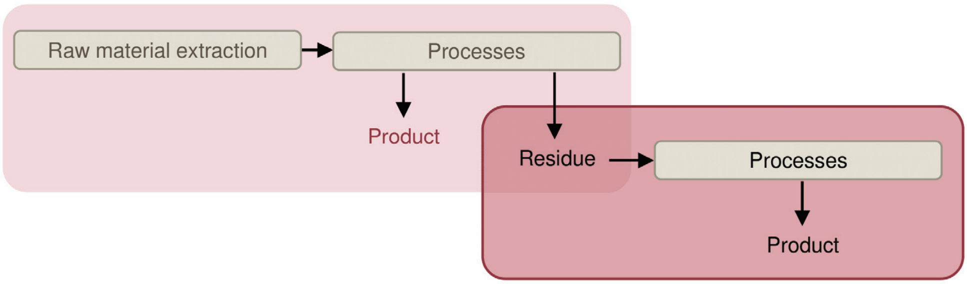


**Figure 2**
Simplified schematic illustration of a residual material as originating from one product system and entering another as a resource, based on Paper II. It is in the delineation between these product systems that the upstream processes of residues as resources must be considered.

The identified studies within the seven sectors dealt differently with by-products and residues, but some common themes could be distinguished. As for the practice of allocating environmental impact to residues, an appropriate basis was one that was considered to reflect an underlying cause of the whole production system, and also relevant characteristics of the residual stream. The underlying cause of production could be viewed either as a demand for nutrients or energy, or an economic demand. This line of reasoning was thus used in favour of both physical allocation based on different types of mass, nutrient contents, and energy, as well as for allocation based on the economic values of outputs. Similarly, methods were disqualified by LCA practitioners if they appeared to fail to fairly depict the dynamics of the industrial sectors and their markets. For instance, the argument that it is not reasonable to allocate a larger share of environmental impacts to residues than to products was seen in several studies, and this problem was mainly attributed to mass-based allocation. On the other hand, the allocation of impacts to residues was considered important to illustrate the contribution of residues (as resources) to unsustainable practices. As an example, some authors argued that if fishing causes the collapse of a fish stock, by-products thereof could not be considered to have a low environmental impact even though they may have a low economic value.

The reasons for choosing a certain method to deal with multi-functionality seems to be based on the specific situation and residue more than on a general underlying rationale for the LCA method. For instance, the idea of basing allocation on valuable characteristics of the residual stream or streams was evident. Authors who applied substitution more often considered the value of the residual material in its consequent application as a basis for dealing with multi-functionality. Allocation methods were instead more often based on the valuable characteristics shared by main and by-products. There was, however, also evidence of a general line of reasoning that low environmental impact allocated to residues could incentivise their further use and valorisation.

Regardless of how the upstream processes related to biomass residues are included, Paper II concludes that the environmental impacts related to primary production reflect on the residues. As an example, in Paper IV, the assessment of two case studies – ethylene from wheat straw and jet fuel from animal by-products – include the climate impacts of upstream processes by economic allocation. The results show that the greenhouse gas emissions related to wheat cultivation, 10% of which are allocated to straw, have little importance to the resulting climate impact compared to other processes. Other field activities related to the removal of straw, including collection and transportation, and fertilisation to compensate for nutrient removal, result in greater greenhouse gas emissions. In the case of animal by-products, however, the 4% of greenhouse gas emissions from animal husbandry that are allocated to the by-products at slaughter have a significant impact on the results. Animal husbandry can of course result in different climate and other impacts, depending on the animals in question; what they are fed, and how they are kept. The case in Paper IV assumes animal by-products from cattle bred for dairy products or beef. Notably, the results indicate that the jet fuel produced from cattle by-products does not necessarily result in lower greenhouse gas emissions than fossil fuels. Others have indicated similar conclusions (Capaz et al., 2020, Seber et al., 2014). The choice of the animal husbandry data, however, is important to the outcome, and can be understood as a type of variability as discussed in chapter 3.5. The inventory of upstream impacts of biomass residues in Paper II showed similar variation between different studies. This may be explained by actual differences between upstream processes, as in the case of animal husbandry above, and by the application of different allocation methods.

## 4.3 Values and intersubjectivity

The long-standing debate on allocation practices in LCA can be better understood, somewhat, as part of a wider debate on value-judgements in the same field. The presence of value-judgements in LCA is a long-debated topic (Freidberg, 2018). There is general agreement that values are present both in technology assessments in general (Grunwald, 2009) and in LCA specifically, and LCAs of waste management are no exception (Ekvall et al., 2007). There has, however, been disagreement concerning whether or not value-judgements and thereby subjectivity are restricted to certain parts or procedures of the method, and therefore whether or not they can be separated from other objective parts (Hertwich et al., 2000). The process of weighting together results from different environmental impact categories (optional according to ISO 14044) is a typical example where value-choices are undoubtedly present, as it implies the valuation of one type of environmental impact compared to another. Other phases of LCA have, however, also been increasingly accepted as sensitive to values, including the alternatives studied (Bras-Klapwijk, 2003), the formulation of goal and scope and the choice of functional unit, the choice of inventory data, and the approach for dealing with uncertainty (Freidberg, 2018). A relevant question could thus concern whether the presence of values is problematic, and how it affects the use of LCA for knowledge creation purposes.

This can be made clearer by differentiating between different types of values in LCA. Hertwich et al. (2000) distinguish between three types of values based on the work by Shrader-Frechette (1991) on risk assessment. The three types are constitutive values, contextual values, and bias or preference values. *Constitutive* values include the acceptance and adherence to a scientific paradigm or method. *Contextual* values concern "*personal, social, cultural, or philosophical emphasis*" in a judgment (Shrader-Frechette, 1991) and may affect the assumptions made by an LCA practitioner and the choice of one data set over another. *Preference* values reflect what we care about, such as moral values, and *bias* values are preference values which are unacceptable in science (Hertwich et al., 2000) because they are used to produce outcomes that serve one's own purposes (Shrader-Frechette, 1991). Bias values are likely the type of values that come to mind when picturing corrupt LCA practice. As an example, actors in the waste management sector have expressed a view of LCA as telling any story that its commissioner wants to tell (Lazarevic, 2015). An argument can, however, be made towards the scientific value and usefulness of LCA, built on transparency and critical peer review:

> "Like science, LCA can be objective in the sense that arguments for method choice fulfill clear criteria and follow rules deemed as reasonable by the community involved in the development of this tool. It can be objective not in the sense that all arguments or methods have the same standing, but that we can distinguish between valid and invalid arguments." (Hertwich et al., 2000)

I will come back to the discussion on LCA's objectivity in chapter 5.1, but in this context, I focus on the role of the common ground developed in LCA communities. The different rationales and ideas for considering the upstream impacts of residues in Paper II can be understood as based on constitutive and contextual values, and arguments to be made subject to peer-review, scrutiny, and evaluation. The problem of bias values within LCA can thus be solved by making the arguments and rationales for method choices explicit, and the subject of discussion and examination by others. A peer-review process such as that described in the quote above can be understood as a type of intersubjectivity happening between practitioners in the sense that it *"[c]oncerns the relations between people, rather than within them (subjectivity) or beyond them (objectivity or transcendental reality)"* (Calhoun, 2002). I consider it at the core of LCA as a method used to create knowledge. It is, however, not likely to function without a certain level of transparency. Therefore, the lack of information provided regarding allocation of environmental impacts to residues that was identified in Paper II, appears all the more problematic.

The role of intersubjectivity should, however, not necessarily be expected to lead to consensus on method choices and results, and consensus should not necessarily be considered a goal. Instead, when made explicit, it is possible to learn from the different values and other often implicit assumptions related to how one interprets e.g. environmental issues and risks; in other words, different *frames* (Bras-Klapwijk, 1999). With this view, the fact that different rationales and methods to handle multi-functionality are present in the literature, as in Paper II, is not in itself problematic. Instead, I suggest that the potentially problematic aspects include a lack of motivation or argument for the assumptions and method choices made, a lack of illustration of their impacts on results, and an interpretation of the results that is not in line with the rationale that guided the method choices.

## 4.4 Interpreting the upstream impacts of residues as resources

In this chapter, we have seen how the environmental impacts of primary production systems reflect on biomass residues, and how this can be handled practically by allocation in LCA. As shown in Paper IV, there are cases where bio-based products based on residual feedstock are associated with a higher climate impact than a fossil alternative, due to the inclusion of upstream processes. The interpretation of such results is not obvious. If the avoidance of the valorisation process does not avoid the creation of the residual material, is it not still better to make use of it? One answer would be yes, when this is the case, a management perspective that allows for assessing the climate impacts related to different options for handling of the material could be called for, including the diversion of residues from soil or other applications as discussed in chapter three.

To inform long-term strategies, however, this line of reasoning is increasingly questionable. As mentioned in the previous parts of this chapter, a small contribution towards making a production system with high climate impact more passable, is a contribution. The inclusion of upstream processes in assessments can make visible the characteristics of the primary production system from which the residues are collected, and which must be accepted in the future scenario where the valorisation of its residual streams is envisioned. As LCA has historically allowed for identifying and showing unintentional environmental impacts that are not obvious to us, it could in this way serve the same purpose for residues as resources. Another answer to the question whether it is not still better to make use of the residual material is therefore that it depends on for what context and for when we interpret the results.

It is also necessary to point out that already at the focus on biomass residues as resources, or on the comparison of different management options for residues, any scenario in which the residues are not created in the first place is omitted. It might be obvious that it is impossible to identify less environmentally impacting alternatives if these are excluded from a study (Bras-Klapwijk, 2003), but less obvious how to avoid such exclusion. Avoiding zero-burden assumptions can be one step towards opening up for prevention strategies in assessments (Cleary, 2010), as is avoiding functional units based on the management of a certain amount of a residual or waste material (Bernstad Saraiva Schott and Cánovas, 2015). From a broader perspective, the comparison of the prevention of biomass residues production to the utilisation of biomass residues as resources, may be interesting and worth exploring. Such prevention could in some cases result from more efficient production processes, but it could also be difficult to untangle from the demand for other products from the same processes. Again, it is problematic to talk about biomass residues without involving the processes and systems in which they were created.

The EU RED and its assumption of biomass residues as free from the climate impacts of upstream processes can similarly be understood as a specific and narrow take on a complex question. The calculation method of the EU RED is part of a policy instrument, and its methods may be fit for a concrete purpose. A policy instrument should, however, not guide a general understanding of the climate impacts associated with different types of biomass feedstocks, since the political ideas and conclusions that it represents are relevant and valid under specific circumstances. This discussion, and similar questions related to when methods and models can be considered useful depending on the purpose and context, are further explored in the next chapter.

# 5 Assessing valorisation towards different goals

When the United Nations' intergovernmental panel on climate change, IPCC, released its first report in 1990, it contained an explanation of the complexity in producing a method to weigh together the climate impacts of different substances into a single number. In the same context, it also gave an example of such a method: the global warming potential, GWP. Despite the IPCC's expression of a lack of scientific preference for this particular method (Myhre et al., 2013b), the GWP has since become the standard characterisation method for climate impact in LCA. Since then, the GWP factors for different climate-impacting compounds and different time-horizons have been updated in more recent IPCC reports, and other methods have been introduced, such as the global temperature change potential, GTP. While both methods aim at quantifying the impact of different substances on the climate, they measure different things and may therefore relate to and inform different types of climate-change mitigation strategies. The practice of weighing together the climate impacts from different compounds as is standard in LCA thus requires a choice of methods which involves value-judgements (Levasseur et al., 2016).

This chapter is focused on the coherence between applied methods and the goals which they are intended to advance. The previous chapters have illustrated how several aspects have implications for the conclusions that can be drawn concerning valorisation of biomass residues and the resulting climate impact, including the chosen route for biomass residues and what it is compared to, the design of conversion processes, and the way in which these are studied using LCA. In this chapter, the role of the climate impact assessment method, or the characterisation method for climate impact, is further explored in the context of residual biomass valorisation. Unlike the climate targets of the Paris Agreement and similar agreements and laws, the goals of circular bioeconomies are not as well defined nor studied, even though the presence of the concept in policy is tangible. In this chapter, the potential of circularity assessment to guide the management of biomass residues towards circular bioeconomies is therefore also explored. The potential contribution from LCA and circularity assessment to guide residual biomass valorisation towards these two goals is lastly discussed from an understanding of imperfect, but useful, models.

## 5.1 Climate impact assessment

The overarching research interest in this work is in climate impacts. While climate impacts have been studied in Papers I, III, and IV, the methods for doing so have varied. In Papers I and IV, the common GWP100 (GWP with a 100-year time horizon) is applied. This characterisation method is, as stated above, in no way scientifically superior to others. However, as it is so widely used to assess climate impacts, applying it allows for comparing the results to many those of many others. For instance, a comparison is made to results following the RED method – which applies GWP100 – in Paper I. Contrastingly, the approach in Paper III includes a comparison of different characterisation methods. The climate impact assessment methods applied in Paper III include GWP and GTP and their time-dependent counterparts AGWP and AGTP (absolute GWP and GTP, respectively). The application of AGWP and AGTP is made possible by life-cycle inventories with a temporal resolution as mentioned in chapter 3.4. When the life-cycle inventory is set up with a temporal resolution, as in Paper III, it is possible to also illustrate a temporal resolution in the results. The results from a dynamic LCA can otherwise be managed as for any LCA with time-aggregated environmental impacts and, in such cases, the characterisation method makes use of dynamic characterisation factors (Levasseur et al., 2010). In Paper III, calculations are instead made where the characterisation method preserves the temporal resolution of the life-cycle inventory.

As established in chapter three, the comparison between the functionally equivalent biofuel and reference scenarios for logging residues is affected by method choices. The importance of the method chosen for including biogenic carbon to the comparison of scenarios for logging residues was discussed in chapter three, but Paper III also shows that the choice of climate impact assessment method can further affect the conclusions. In general, the illustration of climate impact over time shows an initially higher impact in the biofuel scenario, which eventually levels and drops below that of the reference scenario at a certain parity time (Figure 3). The parity times for the biofuel scenario for logging residues in Paper III range between 50 and 95 years with AGWP, and between 25 and 30 years with AGTP. As a reminder, these parity times indicate when the climate impact of the option to harvest and valorise logging residues from Swedish spruce forest to produce energy carriers, levels with that of the reference scenarios in which residues are not harvested and fossil energy carriers are used instead. The ranges describe the difference in parity time between low and high cases which differ e.g. in terms of the assumed greenhouse gas emissions of fossil fuels.

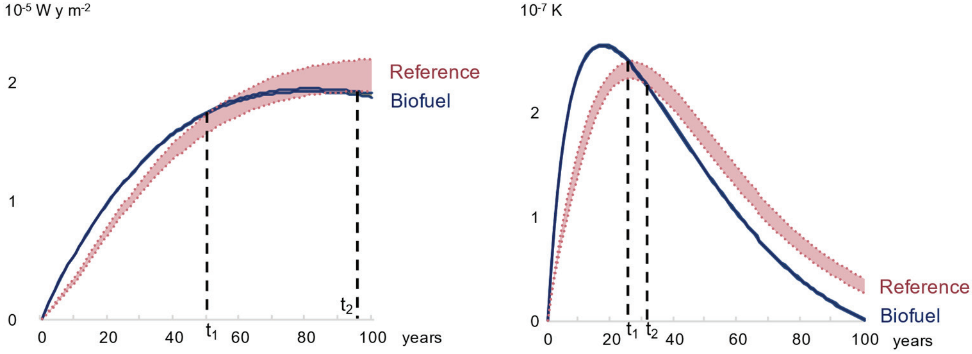


**Figure 3**
Climate impact of the biofuel and reference scenarios for logging residues in Paper III. To the left, the climate impact is calculated with the AGWP method, and to the right, the AGTP method. The vertical dashed lines indicate the parity time, a range between $t_1$ and $t_2$, at which the climate impact of the valorisation scenario levels with that of the reference scenario.

There is a significant difference between the results and parity times calculated with the AGWP and the AGTP method due to differences between these methods. A first difference lies in what is measured: AGWP describes a change in radiative forcing which is a measure of the change in the Earth's energy balance resulting from e.g. an emission of a greenhouse gas (Myhre et al., 2013a). AGTP, on the other hand, describes a change in the global surface mean temperature. Both are midpoint indicators, but compared to the AGWP, the AGTP's measure of temperature change is one step further down the cause-effect chain of climate change (Fuglestvedt et al., 2003). Another difference lies in the cumulative and instantaneous consideration of what is being measured: AGWP is cumulative and describes the sum of radiative forcing up until a certain point in time, and AGTP is instantaneous in the sense that it describes a temperature change at a certain point in time, regardless of the temperature trajectory up until that point. Whether or not the valorisation of logging residues as studied in Paper III, and implicitly also in Paper I, contribute to a climate-change mitigation strategy is therefore a question of what type of climate impact is to be lowered, and when.

The chosen climate impact assessment method also affects the impact of other data and method choices on the comparison between the biofuel and reference scenario. This is illustrated by the wider range of parity times calculated with the AGWP compared to the AGTP, and where the ranges are largely explained by the different assumptions used for the greenhouse gas emissions from fossil energy carriers. In this way, the combination of different methods and scenarios in Paper III further illustrates the importance of different method choices in combination. Contrastingly, the same investigation for the valorisation of Swedish wheat straw in Paper III resulted in immediate climate benefits compared to the reference scenario.

### 5.1.1 Understanding method choices

The application of different methods to assess climate impacts can be viewed as one way to better understand the climate impact of bio-based products, and thereby allow for better informed decisions (Røyne et al., 2016). Cooper et al. (2020) point out that a temporal resolution of results makes it possible for the LCA practitioner to assess how all relevant aspects of the results can best be communicated. Different climate impact assessment methods can also be interpreted as representing different types of environmental impacts. The GWP has been argued a poor choice for representing long-term climate impact (Levasseur et al., 2016, Jolliet et al., 2018) due to its cumulative nature. With this understanding of short and long-term effects on the climate, the general difference between the biofuel and reference scenario for logging residues in Paper III can be more easily observed. The results with the cumulative metrics (GWP, AGWP) favour the reference scenario and result in longer parity times, whereas the results with the instantaneous metrics (GTP, AGTP) favour the biofuel scenario. Interpreting these results as short and long-term impacts, respectively, better correlates with the initially greater impact of the biofuel scenario as can be observed with the results displayed over time, than does the choice of time horizon as 20 or 100 years. This is not to say that either of the methods are better. Rather, "*[a]ny preference of one metric over another arguably favours the representation of some aspects of the climate system response and at the same time discount others*" (Cherubini et al., 2016). Understanding different methods may, however, alleviate *relevance uncertainty* related to how representative an indicator is of the problem we want to mitigate (Rosenbaum et al., 2018a).

The outcome of Paper III is a result of the timing of greenhouse gas emissions from the different activities in the biofuel and the reference scenarios, which include fluxes of carbon atoms of both fossil and biogenic origin. It is sometimes argued that the atmosphere does not care whether the carbon is of fossil origin or not. However, not all greenhouse gas emissions are equal, and not only due to their most recent origin. Cherubini et al. (2016) explain the nature of GWP as indicating that all greenhouse gas emissions are exchangeable – one can pick and choose the necessary amount of a certain greenhouse gas to achieve a sought reduction in GWP. In fact, however, emissions of different greenhouse gases at different points in time lead to different climate impacts. If we are to limit the global average temperature increase, cumulative $CO_2$ emissions such as those from fossil fuels simply must be cut (Cherubini et al., 2016, Allen, 2015). The comparison of the biofuel to a reference scenario with fossil fuels in Paper III thus has its limitations. The representations of the climate impact from the biofuel scenario can nonetheless indicate its temporal dimension and increase our understanding of it. In the end, what is acceptable in terms of temporary warming to achieve long-term emission reductions is a matter both of what else happens in the world, and of explicit climate-change mitigation goals (Allen, 2015).

### 5.1.2 Values in climate impact assessment

With all the choices having to be made to conduct the climate impact assessment, the value debate in the LCA literature that was introduced in chapter four applies here as well. The current version of ISO 14044 acknowledges the presence of value-choices in both the phase of goal and scope definition and in the choice of impact categories and characterisation methods. It states that one should be transparent about and report them, and related to characterisation methods, that one should minimise them (ISO, 2006b). The previous pages, however, point to the need for plurality in climate impact assessment to meet the many facets of the climate-impact mitigation challenge. In addition, a more general critique is that subjectivity is not necessarily bad. It "*does not mean the absence of any scientific guidance of how to perform the necessary discussion about the values needed for performing the "subjective" parts of an LCA*" (Klöpffer, 1998). Freidberg (2018) argues that making explicit the "view from somewhere" or rather someone, as opposed to the objective "view from nowhere", is not only more honest and rational, but also more credible and may aid in building public trust in the practice of LCA.

Similarly, Bras-Klapwijk (1998) identifies the perceived objectivity and authority of LCA as the more problematic aspect of the method in decision-making processes as it effectively closes down on the potential framings and interpretations of environmental issues and risks. A better understanding and communication of the inherent value-choices in applied climate impact assessment methods may therefore help to illustrate the complexity in comparing options, e.g. from a climate-change mitigation point of view. It could also be considered a means to "open up" the results of assessments in the sense that they are communicated "*in a more 'plural and conditional' fashion with respect of whatever are the most salient axes of sensitivity that emerge in any of the input dimensions*" (Ely et al., 2014). With such an approach, final decisions or recommendations are not part of assessments, but instead a broader basis for decisions are provided, and a final decision is effectively left to the decision-maker (Stirling, 2010). This does not mean that any choice or framing is as good as the other, and that no decisions should be made by the LCA practitioner. Rather, it implies that important and decisive framings and choices are made visible, and the knowledgeable LCA practitioner has a role in identifying and illustrating them to allow for discussion and learning.

The interpretation of the scenario comparison in Paper III as put forward in this chapter is quite far from the calculation rules and emission reduction targets for liquid biofuels set up in the EU RED. The RED method assumes the use of GWP100 factors and – at the time of writing – sets the bar for biofuels produced in new installations (since 2021) at a maximum climate impact of 33g $CO_2$-eq. per MJ, which corresponds to 65% reduction compared to a fossil reference set to 94g $CO_2$-eq. per MJ (Article 29, 10.c). It also considers a certain set of biomass residues as burden-free at the point of collection, regardless of their value or alternative use. It is obvious that many value-laden and uncertain choices have already been made in the definition of the EU RED approach. It does not seem too far-fetched, therefore, to argue that such an approach effectively "closes down" in the sense that it aims to determine a best option, rather than "opens up" by illustrating how such a conclusion may be affected by different framings and assumptions (Stirling, 2008).

The RED method, and implicitly the value-judgements made to define it, have been decided upon to promote certain types of feedstocks and biofuels. It has guided the production and consumption of liquid biofuels, and has been important to biofuel producers and the way they approach improvements to biofuels' sustainability (Lazarevic and Martin, 2018). Is that then problematic? I would argue that whether the underlying assumptions and value-judgements are acceptable depends on the intention and application of the RED, as discussed in chapter four. However, acknowledging the power that lies in the RED method to shape European biofuel production, it seems advisable to at least be cautious about what this type of model, or tool, is and does, and what it is not and does not do. I come back to this discussion in chapter 5.3.

## 5.2 Circularity assessment

Compared to the field of LCA and climate impact assessment, circularity assessment is a recent endeavour under development. The circular bioeconomies which encompass the circular resource systems and loops are intended to contribute towards climate-change mitigation, and therefore, climate impact assessment is also highly relevant in this context. However, to better measure bio-based circularity specifically, and to advance the closing of material cycles in different contexts, different indicators and methods for circularity have been suggested. This chapter continues the introduction of circularity assessment as presented in chapters one and two of this thesis and presents findings from the cases studied in Paper IV.

Many methods for measuring circularity are available, but few aim to assess circularity for biomass or bio-based products. Among the methods that have previously been suggested to cover important aspects of bio-based circularity (Navare et al., 2021, Jerome et al., 2022), one was chosen for the analysis of bio-based products from biomass residues in Paper IV. This is the material circularity indicator, MCI, created by the Ellen MacArthur Foundation and ANSYS Granta (2019). Choosing to focus on one single indicator may seem a contrast to the approach to climate impact assessment in this chapter. Indeed, the assessment of the circularity concept is likely to benefit from a wider range of methods and viewpoints, but the method chosen for Paper IV can be seen as representing the current state of circularity assessment for bio-based products. This is because the MCI appears to be one of the most widely applied and assessed product circularity metrics (Bracquené et al., 2020, Rigamonti and Mancini, 2021), and because several of the identified metrics for bio-based circularity were either based on the MCI, or similar to it (Paper IV).

The MCI is calculated on the product level from the fraction of virgin and reused, recycled, or sustainably harvested biomass materials that make up a product's mass, and the fraction of that mass which is reused, recycled, or composted after the use of the product (Ellen MacArthur Foundation and ANSYS Granta, 2019). Additionally, it considers waste and material losses in recycling processes. It also considers the utility of the product compared to an industry standard, which is relevant e.g. to life extension strategies, but this aspect was not considered relevant to the cases studied in Paper IV. The MCI takes a value between 0.1 and 1, where 1 implies a perfectly circular product, and 0.1 a perfectly linear product.

In Paper IV, the MCI was applied to two case studies: ethylene from wheat straw, and jet fuel from animal by-products. Both cases were assessed in several scenarios where different parameters were varied to illustrate their impact on the result, as in a local sensitivity analysis (see 3.5 Data and uncertainty). Already from the outset, however, it was clear that the scope of the MCI differs widely from that of LCA. For instance, the MCI is based on the mass of a product, and the maximum amount of virgin resources that can be considered equals the mass of the product. This critique has been the basis for suggesting elaborations of the method (Bracquené et al., 2020), but further shortcomings are illustrated in Paper IV. For conversion of biomass such as that of straw to ethylene, the MCI does not consider the process additives which are needed in the conversion process, but which do not end up in the mass of the product. The fact that some of these additives constitute hotspots in terms of climate impacts – e.g. the enzymes mentioned in chapter three – could make this limitation more problematic.

The chosen system boundaries also significantly impact the results in both the studied cases. With an economic allocation of the cultivation stage to straw (10%) and of the animal husbandry stage to animal by-products (4%), the MCI scores decrease from approximately 0.6 to 0.2, and from 0.5 to 0.2, respectively. In the case of animal by-products, the fraction of recycled material is then decided by the composition of the feed. Essentially, the circularity score with the MCI does not reflect the magnitude of the material throughput, as long as materials are considered reused, recycled, or sustainably sourced biomass, and as long as they are reused, recycled or composted after the use phase. In line with this limitation, the circularity result did not reflect different greenhouse gas intensities of the animal husbandry stage, which is assumed to be focused either on dairy-production or on beef. The findings from Paper IV thus add to previous research that finds circularity assessment to be an inadequate tool for minimising environmental impacts (Lonca et al., 2018, Helander et al., 2019, Corona et al., 2019). The inability of circularity assessment methods to quantify environmental impacts that they are not intended to is not necessarily surprising, but an important reminder of their limitations.

Other, perhaps more pressing, limitations of the MCI found in Paper IV concern the inability of the method to address important aspects of circular bioeconomies as they are envisioned today. As mentioned in chapter three, biomass residues are expected to contribute both recirculation of organic matter and nutrients to soil, and feedstock for bio-based products. The recirculation of organic matter and nutrients is considered essential to both restore and to further regenerate the primary production systems upon which circular bioeconomies would critically rely. A use of agricultural biomass residues locally has therefore been argued a priority in striving for circularity (Velasco-Muñoz et al., 2022). Regardless of how restoration and regeneration are achieved, Navare et al. (2021) deem the MCI (as part of a package called Circulytics) as explicitly assessing the sustainable sourcing of biomass. It does so by applying a criterion to primary biomass use, stating that the extraction rate and practice should aim to maximise regeneration of natural systems (Ellen MacArthur Foundation and ANSYS Granta, 2019). The assessment of whether a certain biomass meets this criterion is, however, outside the scope of the MCI parameters and calculations.

Instead, the MCI and similar methods focus on the mass input to a product – but limited to the mass of the product – and the handling of the same mass after the product's use. Concerning the mass output, it is not possible to follow the circulation of individual substances as the MCI is based on general mass flows. Composting of one kg of biomass therefore gives a certain contribution towards circularity regardless of its composition. Taken together, these shortcomings illustrate the inability of circularity assessment with the MCI and similar methods to explicitly address some of the most important aspects of circular bioeconomies, and residual biomass use towards them. This type of potential mismatch between the models and methods used for assessment, and the goals they are expected to advance, naturally leads to questions regarding the fruitfulness of their use and application.

## 5.3 Wrong models

> "Any environmental assessment is necessarily imperfect (Funtowicz and Ravetz 1990; Scheringer 1999). Ultimately, arguments in lifecycle assessment should refer back to LCA's purpose in decision making. An imperfect assessment is more likely than no assessment at all to lead to better-informed decisions." (Hertwich et al., 2000)

What hopes can we put in the models discussed in this thesis to advance our understanding and mitigation of pressing climate issues? First, it may help to clarify what the considered model is, and what we expect from it. I consider the *life-cycle model* used in LCA, or rather the life-cycle *system* defined by a functional unit and system boundaries, a model of reality, a model which is a man-made imitation or representation of reality. The model is inherently flawed as it separates certain physical flows from the rest of the world in an artificial way. Here lies the first issue, as different assumptions and value-judgements are made to delineate the life-cycle system. An example is the life cycle of a biomass residue as a resource, separate from the other products of its processes of origin as discussed in Paper II. The second issue arises in the assessment of its environmental impacts where assumptions and value-judgements are again present, as in the example of climate impact assessment in Paper III. Finnveden (2000) concludes that since it is impossible to decide on an assumption or a value-judgement as the single correct one, or eliminate uncertainty for that matter, it is principally impossible for an LCA to conclude that one option is environmentally superior to another – even if that happens to be the case. Despite efforts, there are cases where the use of LCA has not helped to build a common understanding in the decision-making process, including debates on PVC in the Netherlands (Tukker, 2000, Bras-Klapwijk, 1998) and diapers in the USA (Freidberg, 2013). The current debate on bioenergy in the EU and globally could potentially be added to this list.

Following the quote from Hertwich et al. (2000) above, what we want the model to do can arguably be used as a basis for assessing assumptions and choices as acceptable or not. In other, borrowed, words: "*[e]ssentially, all models are wrong, but some are useful*"

(Box and Draper, 1987). A useful model would, in this context, at least live up to the hypothesis in the above quote that an imperfect assessment provides better guidance than no assessment at all. So what does that kind of assessment look like? In chapter 3.5, uncertainty was presented as stemming from variability, measurements and unrepresentative data, models and choices. The nature of uncertainty was further presented as precision and accuracy, where precision describes reproducibility or spread in results, and accuracy describes distance to a target. In this context of wrong models and decision-support, I would argue that a worst-case scenario would be for LCA results to show precise and inaccurate results. Such results could be interpreted as unanimous and certain about environmental improvements of a solution that may instead lead to the opposite development. In a discussion of the merits of different LCA approaches for decision-support, Plevin et al. (2014a) argue that "[i]*f uncertainty prevents a clear determination of benefits, then it is critically important to convey this finding to policy-makers*". Communicating on uncertainty and sensitivity or ambiguity should thus not be compromised for the sake of illusive precision. Rather, as argued also in previous chapters, it could be possible to learn from such uncertainty, sensitivity and complexity.

The view of LCA as expressed above is based on its potential as a model and method to increase our knowledge of the world and thereby include environmental concerns in decision-making processes. There is, however, to some extent, a wish or request for results from LCA-like methods to directly decide for us, which is in contrast to the view of methods such as LCA to advise but not decide (Grunwald, 2009). The EU RED method for liquid biofuels is one such example where a decision has been made to prioritise certain biomass residues over other types of biomass, and measure the contribution to climate-change mitigation in a certain way. Similar lines of thought have been expressed in LCA studies – that LCA results could or should incentivise the use of residual streams by illustrating their lower climate impacts (Paper II, chapter 4.2).

These views are not compatible. The first aims at the accounting of physical flows for a better understanding of the world, and of consequences of decisions. The other aims at the accounting of physical flows from a predetermined set of values to achieve a certain behaviour. Both approaches can be valid and may exist in parallel, but not for the same purpose, and they should therefore not be mixed up. A life-cycle perspective for a better understanding of biomass residues as resources should be allowed complexity where it is fit for the purpose. A policy tool aimed at directly supporting certain processes, as the EU RED, is something different. This type of tension and contesting ideas of how to best develop LCA practice has been observed before (Bras-Klapwijk, 1999), and still remains. With an increasing attention in policy to the circular bioeconomy concept, it is possible that the development and use of circularity assessment in both research and policy contexts will also face similar challenges and distinctions.

Comparing the merits of LCA and circularity assessment in this context may seem unfair, considering the decades of effort put into developing, testing and understanding LCA. Even so, it is evident that current circularity assessment methods do not encompass the critical aspects of circular bioeconomies, and should thus not be taken as measurements of it. Rigamonti and Mancini (2021) suggest that circularity results may be more easily communicated towards different stakeholders, essentially due to the methods being less complex than full LCAs, which include several environmental impact categories, and possibly complexity in other areas. Based on the findings in Paper IV, it seems that such perceived simplicity is hinged on a simplified manifestation of complex systems, and on a narrow definition of circularity. This type of circularity assessment could therefore possibly provide easily interpreted rankings based on a number of decisions and value-judgements, similar to the early use of LCA. However, it could not help to navigate the complexity of biomass use and resource loops in circular bioeconomies.

The models used to assess the climate impacts and circularity of residual biomass use are wrong in the sense that they are imperfect representations of reality, as are all models, in some sense. That does not, however, mean that they cannot be useful. In terms of circularity assessment with the MCI, I argue that the method risks leading decisions in the wrong direction, in the sense that it does not meet the requirement that an imperfect assessment is better than no assessment at all. For LCA studies of biomass residues as resources, I argue that it is the way in which the model and method is used that determines their usefulness.

# 6 Concluding discussion and future research leads

Three perspectives on biomass residues as resources have been explored in the previous chapters: factors that affect a comparison of residue valorisation and other management options, considering the upstream environmental impacts of residues, and the assessment of residual biomass valorisation towards goals of climate-change mitigation and circular bioeconomies. In this concluding chapter, I briefly discuss the findings from each of the previous chapters in relation to the research questions.

First, the idea of biomass residues as inherently better feedstocks for bio-based products than other types of biomass can be challenged. As for biomass in general, different types of biomass residues and valorisation processes provide different opportunities for climate-change mitigation. In addition, the models and methods used to assess the climate impacts related to valorisation and other processes, affect the outcomes.

**How do different factors influence conclusions and comparisons of the climate impacts of different alternatives of biomass residues' management?**

The biomass residues and valorisation processes studied in this thesis show that several choices and aspects related to the LCA method may affect the conclusions drawn for valorisation of biomass residues as part of a climate-change mitigation strategy. First, several aspects affect the resulting greenhouse gas emissions of different management alternatives. These include the process inputs for conversion of lignocellulosic biomass, including the resources and energy carriers used in the production of enzymes; the assumed emission intensity of fossil fuels, both as comparators to bio-based products and as inputs to their production; and the inclusion of climate impacts from upstream processes. Second, the method chosen for assessing the impact of greenhouse gas emissions on the climate, and the consideration of different climate-change mitigation goals, can influence conclusions. Additionally, the methods chosen for considering biogenic $CO_2$ can influence both comparisons of greenhouse gas balances and comparisons of their climate impacts, especially for forest residual biomass. All these factors to some extent influence how sensitive the conclusions are to other factors.

Therefore, the influence of different factors on the conclusions and comparisons of biomass residues' management as studied in this thesis, can be seen as interconnected.

Some of the abovementioned factors stem from incomplete knowledge of the processes involved, such as the contribution of residual biomass to long-term carbon storage in soils. A natural suggestion would be for further research on such aspects of residual biomass utilisation that we have a limited understanding of, and their application in LCA. Several factors and assumptions are, however, also subject to uncertainty about future developments. For instance, the long-term carbon storage in soils may be affected by climate change. Technological and societal development affects the specific numerical values that are relevant to assessments, but also which products and processes it is relevant to compare bio-based products to. Here, LCA studies can have an important role in illustrating the climate impacts of different relevant scenarios and their sensitive parameters, based on available knowledge.

The choice of the goal towards which residual biomass management should be measured adds an additional layer of complexity. The example of logging residues from Swedish forestry shows how LCA can quantify and illustrate the climate impacts of biomass residues' valorisation in different ways, and how the conclusion regarding climate-change mitigation potential can depend on when and in what sense the climate impact is considered. In this context, the application of different climate impact assessment methods can help to illustrate the choices at hand, and potentially provide the basis for a more nuanced discussion of when and how biomass residues can be useful as part of climate-change mitigation strategies. In some cases, the choice of climate impact assessment method can also affect how sensitive the conclusions are to other parameters, such as the assumed emission intensity of fossil fuels.

Many factors other than those included here could further influence results and conclusions, both within climate impact assessment methods and otherwise. This thesis thus points to some potentially important factors that should be considered, while there are certainly more to explore.

**How can the impacts of primary biomass production and other processes in which biomass residues are created be considered in assessments? What implications may such consideration have to conclusions for residual biomass valorisation?**

Assumptions of residues as free from the climate and environmental impacts of primary production systems are to some extent already outdated, and will increasingly become so, in more circular resource systems with higher value of biomass residues. Studies of bio-based products made from biomass residues should therefore increasingly include upstream processes and their impacts. The literature indicates that an allocation of upstream impacts to residues that reflects the main driver or cause of production, is seen as reasonable by many LCA practitioners. Whether such a driver or cause is best described as an economic demand, or a demand for e.g. nutrients or energy, is a type of value-judgement that is likely to remain a part of LCA practice, calling for transparency and intersubjectivity.

The inclusion of upstream processes in assessments of residual biomass valorisation can potentially have a significant impact on the climate impacts of bio-based products. In cases where the resulting climate impact of the bio-based product is in the order of, or greater than, that of a fossil-based product, results can be interpreted with different perspectives in mind. Essentially, the valorisation of residues can be seen as a support for making or keeping the primary production systems which the residues come from more viable. In other words, the primary production and its residual streams cannot be considered or assessed in isolation. This may seem an elephant in the room. There are ways in LCA to consider a single product from shared processes, or a residual stream as a resource apart from its origin. This is, however, an analytical exercise to help us understand and consider potential environmental impacts in decision-making processes. This exercise can be valuable as long as it can support decisions towards minimising climate and other environmental impacts. By assuming that residues are burden-free at the point of harvest or collection, one risks jeopardising this important function of LCA. It must, however, be possible to keep two thoughts alive at once, so that increasing attention to valorisation can still be supported in cases where current management of biomass residues leads to environmental issues.

It is also essential, in this context, to distinguish between the different goals of learning and understanding by means of LCA, and of steering behaviours with policy instruments based on LCA methods. The latter is exemplified in this thesis by the calculation approach for biofuels in the EU renewable energy directive, RED, which is influenced by political priorities such as the promotion of residual biomass use for biofuels. Its results should be understood as outcomes and mirrors of those priorities, which may be discussed and may change over time. Here, LCA and other assessment tools may be used to understand the environmental impacts of political priorities and policies, and thereby inform them. The findings of this thesis illustrate how the upstream processes of biomass residues reflect on the climate and environmental impact of bio-based products, and how the inclusion of such upstream processes may significantly influence climate-impact results and conclusions. This leads to questions regarding the political priority to categorically promote residual biomass use for biofuel production. Future strategies should therefore consider the time perspectives and contexts in which different types of biomass residues are promoted as feedstocks for products and fuels.

**In what ways do climate impact assessment within LCA and bio-based circularity assessment provide additional or contrasting insights into residual biomass use?**

The potential dual roles of biomass residues as feedstocks for products on the one hand, and as the bridge to close cycles of nutrients and organic matter on the other hand, are explicitly expressed in visions of circular bioeconomies. Within expressions of these ideas, it is also evident that the long-term productivity and sustainability of primary production systems are the highest priority for viable, bio-based resource systems. The lack of methods coherent with these goals and for explicitly assessing these aspects in currently available product circularity assessment methods can therefore be disappointing, and possibly a result of a focus on qualitatively different materials such as metals, minerals, and plastics, in circular economy research. It is, however, evident that circularity metrics need to be adapted to bio-based products and cycles to be useful. For instance, the efficient use of biomass resources and the use of process additives should be considered for valorisation options. A narrower focus on cycles of different nutrients could, for instance, better support the restorative aspect of circular bioeconomies and thus, potentially, complement LCA. Whether circularity assessments can be useful towards these ends, and more useful than existing methods such as resource use categories in LCA, or material flow analyses, depends on their development.

From a broader perspective, the use of residues towards e.g. soil carbon storage and productivity must be understood as a limited aspect of primary production systems that can, in other ways, be adapted to support long-term health, productivity, resilience, and other valuable characteristics. Here, studies at another scale may be needed. There is already a research interest in bioeconomies at the regional level, which may also be fruitful in this regard.

This latter point of biomass residues as a limited part of bigger systems is central, especially to future strategies, whether related to climate-change mitigation or circular bioeconomies. This thesis has aimed at expanding the life-cycle perspective applied to biomass residues in order to contribute to a use of LCA that can both lead to a better understanding of how biomass residues' management can contribute towards different societal goals, and based on that, provide relevant and useful information to decision-making processes. With that said, if our aim is to limit global warming to a certain degree, or to achieve circular resource systems and sustainable primary production systems, the management of residual biomass must be understood as connected to, and dependent on, the bigger systems in which the residues are created.

# 7 References


Ahlgren, S., Björklund, A., Ekman, A., Karlsson, H., Berlin, J., Börjesson, P., Ekvall, T., Finnveden, G., Janssen, M. & Strid, I. 2015. Review of methodological choices in LCA of biorefinery systems - key issues and recommendations. *Biofuels, Bioproducts and Biorefining,* 9, 606-619.

Albers, A., Collet, P., Benoist, A. & Hélias, A. 2020. Back to the future: dynamic full carbon accounting applied to prospective bioenergy scenarios. *The International Journal of Life Cycle Assessment,* 25, 1242-1258.

Allen, M. 2015. Short-lived promise? The Science and Policy of Cumulative and Short-Lived Climate Pollutants. *Oxford Martin Policy Paper.* Oxford Martin School, University of Oxford.

Andrade Díaz, C., Clivot, H., Albers, A., Zamora-Ledezma, E. & Hamelin, L. 2023. The crop residue conundrum: Maintaining long-term soil organic carbon stocks while reinforcing the bioeconomy, compatible endeavors? *Applied Energy,* 329.

Arvidsson, R., Tillman, A.-M., Sandén, B. A., Janssen, M., Nordelöf, A., Kushnir, D. & Molander, S. 2018. Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. *Journal of Industrial Ecology,* 22, 1286-1294.

Beloin-Saint-Pierre, D., Albers, A., Hélias, A., Tiruta-Barna, L., Fantke, P., Levasseur, A., Benetto, E., Benoist, A. & Collet, P. 2020. Addressing temporal considerations in life cycle assessment. *Science of The Total Environment,* 743, 140700.

Bernstad Saraiva Schott, A. & Cánovas, A. 2015. Current practice, challenges and potential methodological improvements in environmental evaluations of food waste prevention – A discussion paper. *Resources, Conservation and Recycling,* 101, 132-142.

Björklund, A. E. 2002. Survey of Approaches to Improve Reliability in LCA. *The International Journal of Life Cycle Assessment,* 7, 64-72.

Bjørn, A., Owsianiak, M., Molin, C. & Hauschild, M. Z. 2018. LCA History. *In:* Hauschild, M. Z., Rosenbaum, R. K. & Olsen, S. I. (eds.) *Life Cycle Assessment: Theory and Practice.* Cham: Springer International Publishing.

Björnsson, L. & Prade, T. 2021. Sustainable Cereal Straw Management: Use as Feedstock for Emerging Biobased Industries or Cropland Soil Incorporation? *Waste and Biomass Valorization,* 12, 5649-5663.

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,* 33, 308-320.

Bos, H. L. & Broeze, J. 2020. Circular bio-based production systems in the context of current biomass and fossil demand. *Biofuels, Bioproducts and Biorefining,* 14, 187-197.

Box, G. E. P. & Draper, N. R. 1987. *Empirical model-building and response surfaces,* Oxford, England, John Wiley & Sons.

Bracquené, E., Dewulf, W. & Duflou, J. R. 2020. Measuring the performance of more circular complex product supply chains. *Resources, Conservation and Recycling,* 154, 104608.

Brandão, M., Clift, R., Cowie, A. & Greenhalgh, S. 2014. The Use of Life Cycle Assessment in the Support of Robust (Climate) Policy Making: Comment on "Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation …". *Journal of Industrial Ecology,* 18, 461-463.

Brander, M., Burritt, R. L. & Christ, K. L. 2019. Coupling attributional and consequential life cycle assessment: A matter of social responsibility. *Journal of Cleaner Production,* 215, 514-521.

Bras-Klapwijk, R. M. 1998. Are life cycle assessments a threat to sound public policy making? *The International Journal of Life Cycle Assessment,* 3, 333-342.

Bras-Klapwijk, R. M. 1999. *Adjusting Life Cycle Assessment Methodology for Use in Public Policy Discourse.* PhD thesis, Delft University of Technology.

Bras-Klapwijk, R. M. 2003. Procedures and tools for generating and selecting alternatives in LCA. *The International Journal of Life Cycle Assessment,* 8, 266-272.

Bugge, M., Hansen, T. & Klitkou, A. 2016. What Is the Bioeconomy? A Review of the Literature. *Sustainability,* 8, 691.

Calhoun, C. 2002. Intersubjectivity. *In:* Cahoun, C. (ed.) *Dictionary of the Social Sciences.* Oxford University Press.

Capaz, R. S., de Medeiros, E. M., Falco, D. G., Seabra, J. E. A., Osseweijer, P. & Posada, J. A. 2020. Environmental trade-offs of renewable jet fuels in Brazil: Beyond the carbon footprint. *Science of The Total Environment,* 714, 136696.

Carlqvist, K. 2022. *Process development and environmental assessment within softwood based biorefineries.* PhD thesis, Lund University.

Cherubini, F., Bird, N. D., Cowie, A., Jungmeier, G., Schlamadinger, B. & Woess-Gallasch, S. 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. *Resources, Conservation and Recycling,* 53, 434-447.

Cherubini, F., Fuglestvedt, J., Gasser, T., Reisinger, A., Cavalett, O., Huijbregts, M. A. J., Johansson, D. J. A., Jørgensen, S. V., Raugei, M., Schivley, G., Strømman, A. H., Tanaka, K. & Levasseur, A. 2016. Bridging the gap between impact assessment methods and climate science. *Environmental Science & Policy,* 64, 129-140.

Cherubini, F., Peters, G. P., Berntsen, T., Strømman, A. H. & Hertwich, E. 2011a. CO2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. *GCB Bioenergy,* 3, 413-426.

Cherubini, F., Stromman, A. H. & Ulgiati, S. 2011b. Influence of allocation methods on the environmental performance of biorefinery products-A case study. *Resources, Conservation and Recycling,* 55, 1070-1077.

Cleary, J. 2010. The incorporation of waste prevention activities into life cycle assessments of municipal solid waste management systems: methodological issues. *International Journal of Life Cycle Assessment,* 15, 579-589.

Clift, R., Doig, A. & Finnveden, G. 2000. The Application of Life Cycle Assessment to Integrated Solid Waste Management. *Process Safety and Environmental Protection,* 78, 279-287.

Cooper, S. J. G., Green, R., Hattam, L., Röder, M., Welfle, A. & McManus, M. 2020. Exploring temporal aspects of climate-change effects due to bioenergy. *Biomass and Bioenergy,* 142, 105778.

Corona, B., Shen, L., Reike, D., Rosales Carreón, J. & Worrell, E. 2019. Towards sustainable development through the circular economy—A review and critical assessment on current circularity metrics. *Resources, Conservation and Recycling,* 151, 104498.

Creutzig, F., Ravindranath, N. H., Berndes, G., Bolwig, S., Bright, R., Cherubini, F., Chum, H., Corbera, E., Delucchi, M., Faaij, A., Fargione, J., Haberl, H., Heath, G., Lucon, O., Plevin, R., Popp, A., Robledo-Abad, C., Rose, S., Smith, P., Stromman, A., Suh, S. & Masera, O. 2015. Bioenergy and climate change mitigation: an assessment. *GCB Bioenergy,* 7, 916-944.

Djuric Ilic, D., Eriksson, O., Ödlund, L. & Åberg, M. 2018. No zero burden assumption in a circular economy. *Journal of Cleaner Production,* 182, 352-362.

Ekvall, T., Assefa, G., Björklund, A., Eriksson, O. & Finnveden, G. 2007. What life-cycle assessment does and does not do in assessments of waste management. *Waste Management,* 27, 989-996.

Ekvall, T. & Finnveden, G. 2001. Allocation in ISO 14041—a critical review. *Journal of Cleaner Production,* 9, 197-208.

Ekvall, T., Tillman, A.-M. & Molander, S. 2005. Normative ethics and methodology for life cycle assessment. *Journal of Cleaner Production,* 13, 1225-1234.

Ellen MacArthur Foundation and ANSYS Granta 2019. Circularity indicators - An approach to measuring circularity - Methodology.

Elliott, K. C. 2022. A Taxonomy of Transparency in Science. *Canadian Journal of Philosophy,* 52, 342-355.

Ely, A., Van Zwanenberg, P. & Stirling, A. 2014. Broadening out and opening up technology assessment: Approaches to enhance international development, co-ordination and democratisation. *Research Policy,* 43, 505-518.

Energimyndigheten 2022. Drivmedel 2021: Resultat och analys av rapportering enligt regelverken för hållbarhetskriterier, reduktionsplikt och drivmedelslag. Energimyndigheten: Eskilstuna.

Escobar, N. & Laibach, N. 2021. Sustainability check for bio-based technologies: A review of process-based and life cycle approaches. *Renewable and Sustainable Energy Reviews,* 135, 110213.

European Commission 2018. A sustainable bioeconomy for Europe: strengthening the connection between economy, society and the environment. Updated Bioeconomy Strategy. Brussels, European Commission

European Commission. 2023. *The bioeconomy in different countries*. Available: https://knowledge4policy.ec.europa.eu/visualisation/bioeconomy-different-countries_en#ep_natstrat [Accessed 8 March 2023].

European Environment Agency EEA 2018. The circular economy and the bioeconomy Partners in sustainability. Luxembourg, Publications Office of the European Union.

Fargione, J., Hill, J., Tilman, D., Polasky, S. & Hawthorne, P. 2008. Land Clearing and the Biofuel Carbon Debt. *Science,* 319, 1235-1238.

Finnveden, G. 1999. Methodological aspects of life cycle assessment of integrated solid waste management systems. *Resources, Conservation and Recycling,* 26, 173-187.

Finnveden, G. 2000. On the limitations of life cycle assessment and environmental systems analysis tools in general. *The International Journal of Life Cycle Assessment,* 5, 229.

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinee, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D. & Suh, S. 2009. Recent developments in Life Cycle Assessment. *Journal of Environmental Management,* 91, 1-21.

Finnveden, G. & Moberg, Å. 2005. Environmental systems analysis tools – an overview. *Journal of Cleaner Production,* 13, 1165-1173.

Freidberg, S. 2013. Calculating sustainability in supply chain capitalism. *Economy and Society,* 42, 571-596.

Freidberg, S. 2018. From behind the curtain: talking about values in LCA. *The International Journal of Life Cycle Assessment,* 23, 1410-1414.

Fuglestvedt, J. S., Berntsen, T. K., Godal, O., Sausen, R., Shine, K. P. & Skodvin, T. 2003. Metrics of Climate Change: Assessing Radiative Forcing and Emission Indices. *Climatic Change,* 58, 267-331.

Geissdoerfer, M., Savaget, P., Bocken, N. M. P. & Hultink, E. J. 2017. The Circular Economy – A new sustainability paradigm? *Journal of Cleaner Production,* 143, 757-768.

Ghosh, A. 2015. Thinking in Systems. *In:* Ghosh, A. (ed.) *Dynamic Systems for Everyone: Understanding How Our World Works.* Cham: Springer International Publishing.

Gilpin, G. S. & Andrae, A. S. G. 2017. Comparative attributional life cycle assessment of European cellulase enzyme production for use in second-generation lignocellulosic bioethanol production. *The International Journal of Life Cycle Assessment,* 22, 1034-1053.

Giuntoli, J., Ramcilovic-Suominen, S., Oliver, T., Kallis, G., Monbiot, G. & Mubareka, S. 2023. *Exploring new visions for a sustainable bioeconomy,* Luxembourg, Publications Office of the European Union.

Giurca, A. & Befort, N. 2023. Deconstructing substitution narratives: The case of bioeconomy innovations from the forest-based sector. *Ecological Economics,* 207, 107753.

Graedel, T. E. & Allenby, B. R. 1995. *Industrial ecology,* Englewood Cliffs, London, Prentice Hall, Prentice-Hall International.

Graham, R. L., Nelson, R., Sheehan, J., Perlack, R. D. & Wright, L. L. 2007. Current and Potential U.S. Corn Stover Supplies. *Agronomy Journal,* 99, 1-11.

Grunwald, A. 2009. Technology Assessment: Concepts and Methods. *In:* Meijers, A. (ed.) *Philosophy of Technology and Engineering Sciences.* Amsterdam: North-Holland.

Guest, G., Cherubini, F. & Strømman, A. H. 2013. The role of forest residues in the accounting for the global warming potential of bioenergy. *GCB Bioenergy,* 5, 459-466.

Guinée, J. B., Heijungs, R. & Huppes, G. 2004. Economic allocation: Examples and derived decision tree. *International Journal of Life Cycle Assessment,* 9, 23.

Gustavsson, L., Haus, S., Ortiz, C. A., Sathre, R. & Truong, N. L. 2015. Climate effects of bioenergy from forest residues in comparison to fossil energy. *Applied Energy,* 138, 36-50.

Hadley Kershaw, E., Hartley, S., McLeod, C. & Polson, P. 2021. The Sustainable Path to a Circular Bioeconomy. *Trends in Biotechnology,* 39, 542-545.

Hanssen, S. V. & Huijbregts, M. A. J. 2019. Assessing the environmental benefits of utilising residual flows. *Resources, Conservation and Recycling,* 150, 104433.

Helander, H., Petit-Boix, A., Leipold, S. & Bringezu, S. 2019. How to monitor environmental pressures of a circular economy: An assessment of indicators. *Journal of Industrial Ecology,* 23, 1278-1291.

Hertwich, E. G., Hammitt, J. K. & Pease, W. S. 2000. A Theoretical Foundation for Life-Cycle Assessment. *Journal of Industrial Ecology,* 4, 13-28.

Huijbregts, M. A. J. 1998. Application of uncertainty and variability in LCA. *The International Journal of Life Cycle Assessment,* 3, 273-280.

Huijbregts, M. A. J., Gilijamse, W., Ragas, A. M. J. & Reijnders, L. 2003. Evaluating Uncertainty in Environmental Life-Cycle Assessment. A Case Study Comparing Two Insulation Options for a Dutch One-Family Dwelling. *Environmental Science & Technology,* 37, 2600-2608.

ISO 2006a. Environmental management - Life cycle assessment - Principles and framework (ISO 14040:2006). International Organization for Standardization.

ISO 2006b. Environmental management - Life cycle assessment - Requirements and guidelines (ISO 14044:2006). International Organization for Standardization.

Jerome, A., Helander, H., Ljunggren, M. & Janssen, M. 2022. Mapping and testing circular economy product-level indicators: A critical review. *Resources, Conservation and Recycling,* 178, 106080.

Jolliet, O., Antón, A., Boulay, A.-M., Cherubini, F., Fantke, P., Levasseur, A., McKone, T. E., Michelsen, O., Milà i Canals, L., Motoshita, M., Pfister, S., Verones, F., Vigon, B. & Frischknecht, R. 2018. Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use. *The International Journal of Life Cycle Assessment,* 23, 2189-2207.

Kardung, M., Cingiz, K., Costenoble, O., Delahaye, R., Heijman, W., Lovrić, M., van Leeuwen, M., M'Barek, R., van Meijl, H., Piotrowski, S., Ronzon, T., Sauer, J., Verhoog, D., Verkerk, P. J., Vrachioli, M., Wesseler, J. H. H. & Zhu, B. X. 2021. Development of the Circular Bioeconomy: Drivers and Indicators. *Sustainability,* 13, 413.

Karlsson, H. 2018. *Climate impact and energy balance of emerging biorefinery systems.* PhD thesis, Swedish University of Agricultural Sciences (SLU).

Karlsson, H., Ahlgren, S., Sandgren, M., Passoth, V., Wallberg, O. & Hansson, P.-A. 2017. Greenhouse gas performance of biochemical biodiesel production from straw: soil organic carbon changes and time-dependent climate impact. *Biotechnology for Biofuels,* 10, 217.

Kirchherr, J., Reike, D. & Hekkert, M. 2017. Conceptualizing the circular economy: An analysis of 114 definitions. *Resources, Conservation and Recycling,* 127, 221-232.

Klein, O., Nier, S. & Tamásy, C. 2022. Towards a Circular Bioeconomy? Pathways and Spatialities of Agri-Food Waste Valorisation. *Tijdschrift voor Economische en Sociale Geografie,* 113, 194-210.

Klöpffer, W. 1998. Subjective is not arbitrary. *The International Journal of Life Cycle Assessment,* 3, 61-62.

Koponen, K., Soimakallio, S., Kline, K. L., Cowie, A. & Brandão, M. 2018. Quantifying the climate effects of bioenergy – Choice of reference system. *Renewable and Sustainable Energy Reviews,* 81, 2271-2280.

Korhonen, J., Honkasalo, A. & Seppälä, J. 2018. Circular Economy: The Concept and its Limitations. *Ecological Economics,* 143, 37-46.

Lal, R. 2005. World crop residues production and implications of its use as a biofuel. *Environment International,* 31, 575-584.

Lamers, P. & Junginger, M. 2013. The 'debt' is in the detail: A synthesis of recent temporal forest carbon analyses on woody biomass for energy. *Biofuels, Bioproducts and Biorefining,* 7, 373-385.

Lazarevic, D. 2015. The legitimacy of life cycle assessment in the waste management sector. *The International Journal of Life Cycle Assessment*.

Lazarevic, D. & Martin, M. 2018. Life cycle assessment calculative practices in the Swedish biofuel sector: Governing biofuel sustainability by standards and numbers. *Business Strategy and the Environment,* 27, 1558-1568.

Lazarevic, D. & Valve, H. 2017. Narrating expectations for the circular economy: Towards a common and contested European transition. *Energy Research & Social Science,* 31, 60-69.

Levasseur, A., Cavalett, O., Fuglestvedt, J. S., Gasser, T., Johansson, D. J. A., Jørgensen, S. V., Raugei, M., Reisinger, A., Schivley, G., Strømman, A., Tanaka, K. & Cherubini, F. 2016. Enhancing life cycle impact assessment from climate science: Review of recent findings and recommendations for application to LCA. *Ecological Indicators,* 71, 163-174.

Levasseur, A., Lesage, P., Margni, M., Deschênes, L. & Samson, R. 2010. Considering Time in LCA: Dynamic LCA and Its Application to Global Warming Impact Assessments. *Environmental Science & Technology,* 44, 3169-3174.

Lifset, R. J. 2006. Industrial Ecology and Life Cycle Assessment: What´s the Use? *The International Journal of Life Cycle Assessment,* 11, 14-16.

Lindholm, E.-L., Stendahl, J., Berg, S. & Hansson, P.-A. 2011. Greenhouse gas balance of harvesting stumps and logging residues for energy in Sweden. *Scandinavian Journal of Forest Research,* 26, 586-594.

Lonca, G., Muggéo, R., Imbeault-Tétreault, H., Bernard, S. & Margni, M. 2018. Does material circularity rhyme with environmental efficiency? Case studies on used tires. *Journal of Cleaner Production,* 183, 424-435.

Lueddeckens, S., Saling, P. & Guenther, E. 2020. Temporal issues in life cycle assessment—a systematic review. *The International Journal of Life Cycle Assessment,* 25, 1385-1401.

MacLean, H. L. & Spatari, S. 2009. The contribution of enzymes and process chemicals to the life cycle of ethanol. *Environmental Research Letters,* 4.

McCormick, K. & Kautto, N. 2013. The Bioeconomy in Europe: An Overview. *Sustainability,* 5, 2589-2608.

McManus, M. C., Taylor, C. M., Mohr, A., Whittaker, C., Scown, C. D., Borrion, A. L., Glithero, N. J. & Yin, Y. 2015. Challenge clusters facing LCA in environmental decision-making - what we can learn from biofuels. *International Journal of Life Cycle Assessment,* 20, 1399-1414.

Meadows, D. H. 2009. *Thinking in systems: a primer,* London, Earthscan.

Merriam-Webster. n.d.-a. *residue*. Merriam-Webster.com dictionary. Available: https://www.merriam-webster.com/dictionary/residue [Accessed 2023-03-21].

Merriam-Webster. n.d.-b. *waste*. Merriam-Webster.com dictionary. Available: https://www.merriam-webster.com/dictionary/waste [Accessed 2023-03-21].

Muscat, A., de Olde, E. M., Ripoll-Bosch, R., Van Zanten, H. H. E., Metze, T. A. P., Termeer, C. J. A. M., van Ittersum, M. K. & de Boer, I. J. M. 2021. Principles, drivers and opportunities of a circular bioeconomy. *Nature Food,* 2, 561-566.

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T. & Zhang, H. 2013a. Anthropogenic and Natural Radiative Forcing Supplementary Material. *In:* Qin, T. F. S. D., Plattner, G.-K., Tignor, M., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex, V. & Midgley, P. M. (eds.) *The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T. & Zhang, H. 2013b. Anthropogenic and Natural Radiative Forcing. *In:* Stocker, T. F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex, V. & Midgley, P. M. (eds.) *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Navare, K., Muys, B., Vrancken, K. C. & Van Acker, K. 2021. Circular economy monitoring – How to make it apt for biological cycles? *Resources, Conservation and Recycling,* 170, 105563.

Netherlands Enterprise Agency 2019. Guidance on the classification of biomass: categories and NTA 8003 codes under the SDE+ scheme. The Hague.

Oldfield, T. L., White, E. & Holden, N. M. 2018. The implications of stakeholder perspective for LCA of wasted food and green waste. *Journal of Cleaner Production,* 170, 1554-1564.

Oxford English Dictionary. 2022a. *by-product, n.* Oxford University Press. Available: https://www.oed.com/view/Entry/25592?redirectedFrom=by-product [Accessed 2023-03-21].

Oxford English Dictionary. 2022b. *residue, n.* Oxford University Press. Available: https://www.oed.com/view/Entry/163588?rskey=WIwnLn&result=1 [Accessed 2023-03-21].

Oxford English Dictionary. 2022c. *waste, n.* Oxford University Press. Available: https://www.oed.com/view/Entry/226027?rskey=foo7hX&result=1&isAdvanced=false [Accessed 2023-03-21].

Pelletier, N., Ardente, F., Brandão, M., De Camillis, C. & Pennington, D. 2015. Rationales for and limitations of preferred solutions for multi-functionality problems in LCA: is increased consistency possible? *International Journal of Life Cycle Assessment,* 20, 74-86.

Perlack, R., Wright, L., Turhollow, A., Graham, R., Stokes, B. & Erbach, D. 2005. Biomass as Feedstock for A Bioenergy and Bioproducts Industry: The Technical Feasibility of a Billion-Ton Annual Supply. *Biomass as Feedstocks for a Bioenergy and Bioproducts Industry: the Technical Feasibility of a Billion-ton Annual Supply*, 72.

Pfau, S. 2015. Residual Biomass: A Silver Bullet to Ensure a Sustainable Bioeconomy? *The European Conference on Sustainability, Energy & the Environment 2015.* Brighton, United Kingdom: The International Academic Forum (IAFOR).

Pfau, S. 2019. *The Green Choice: Biomass use for a sustainable future.* PhD thesis, Radboud University.

Pichery, C. 2014. Sensitivity Analysis. *In:* Wexler, P. (ed.) *Encyclopedia of Toxicology (Third Edition).* Oxford: Academic Press.

Plevin, R. J., Delucchi, M. a. & Creutzig, F. 2014a. Response to "On the uncanny capabilities of consequential LCA" by Sangwon Suh and Yi Yang (Int J Life Cycle Assess, doi: 10.1007/s11367-014-0739-9). *The International Journal of Life Cycle Assessment,* 19, 1559-1560.

Plevin, R. J., Delucchi, M. A. & Creutzig, F. 2014b. Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. *Journal of Industrial Ecology,* 18, 73-83.

Pradel, M., Aissani, L., Villot, J., Baudez, J. C. & Laforest, V. 2016. From waste to added value product: towards a paradigm shift in life cycle assessment applied to wastewater sludge - a review. *Journal of Cleaner Production,* 131, 60-75.

Ranius, T., Hämäläinen, A., Egnell, G., Olsson, B., Eklöf, K., Stendahl, J., Rudolphi, J., Sténs, A. & Felton, A. 2018. The effects of logging residue extraction for energy on ecosystem services and biodiversity: A synthesis. *Journal of Environmental Management,* 209, 409-425.

Rigamonti, L. & Mancini, E. 2021. Life cycle assessment and circularity indicators. *International Journal of Life Cycle Assessment,* 26, 1937-1942.

Rosenbaum, R. K., Georgiadis, S. & Fantke, P. 2018a. Uncertainty Management and Sensitivity Analysis. *In:* Hauschild, M. Z., Rosenbaum, R. K. & Olsen, S. I. (eds.) *Life Cycle Assessment: Theory and Practice.* Cham: Springer International Publishing.

Rosenbaum, R. K., Hauschild, M. Z., Boulay, A.-M., Fantke, P., Laurent, A., Núñez, M. & Vieira, M. 2018b. Life Cycle Impact Assessment. *In:* Hauschild, M. Z., Rosenbaum, R. K. & Olsen, S. I. (eds.) *Life Cycle Assessment: Theory and Practice.* Cham: Springer International Publishing.

Røyne, F., Peñaloza, D., Sandin, G., Berlin, J. & Svanström, M. 2016. Climate impact assessment in life cycle assessments of forest products: implications of method choice for results and decision-making. *Journal of Cleaner Production,* 116, 90-99.

Schaubroeck, T., Schrijvers, D., Schaubroeck, S., Moretti, C., Zamagni, A., Pelletier, N., Huppes, G. & Brandão, M. 2022. Definition of Product System and Solving Multifunctionality in ISO 14040–14044: Inconsistencies and Proposed Amendments—Toward a More Open and General LCA Framework. *Frontiers in Sustainability,* 3.

Schrijvers, D. L., Loubet, P. & Sonnemann, G. 2016. Critical review of guidelines against a systematic framework with regard to consistency on allocation procedures for recycling in LCA. *International Journal of Life Cycle Assessment,* 21, 994-1008.

Searchinger, T., Heimlich, R., Houghton, R. A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D. & Yu, T.-h. 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. *Science,* 319, 1238-1240.

Seber, G., Malina, R., Pearlson, M. N., Olcay, H., Hileman, J. I. & Barrett, S. R. H. 2014. Environmental and economic assessment of producing hydroprocessed jet and diesel fuel from waste oils and tallow. *Biomass and Bioenergy,* 67, 108-118.

Shrader-Frechette, K. S. 1991. Risk and Rationality - Philosophical Foundations for Populist Reforms. University of California Press.

Silalertruksa, T. & Gheewala, S. H. 2013. A comparative LCA of rice straw utilization for fuels and fertilizer in Thailand. *Bioresource Technology,* 150, 412-419.

Slade, R., Bauen, A. & Shah, N. 2009. The greenhouse gas emissions performance of cellulosic ethanol supply chains in Europe. *Biotechnology for Biofuels,* 2, 15.

Sohn, J., Kalbar, P., Goldstein, B. & Birkved, M. 2020. Defining Temporally Dynamic Life Cycle Assessment: A Review. *Integrated Environmental Assessment and Management,* 16, 314-323.

Starke, J. R., Metze, T. A. P., Candel, J. J. L. & Termeer, C. J. A. M. 2022. Conceptualizing controversies in the EU circular bioeconomy transition. *Ambio*.

Stegmann, P., Londo, M. & Junginger, M. 2020. The circular bioeconomy: Its elements and role in European bioeconomy clusters. *Resources, Conservation & Recycling: X,* 6, 100029.

Stirling, A. 2008. "Opening Up" and "Closing Down": Power, Participation, and Pluralism in the Social Appraisal of Technology. *Science, Technology, & Human Values,* 33, 262-294.

Stirling, A. 2010. Keep it complex. *Nature,* 468, 1029-1031.

Tan, E. C. D. & Lamers, P. 2021. Circular Bioeconomy Concepts—A Perspective. *Frontiers in Sustainability,* 2.

Tonini, D., Hamelin, L. & Astrup, T. F. 2016. Environmental implications of the use of agro-industrial residues for biorefineries: application of a deterministic model for indirect land-use changes. *Global Change Biology Bioenergy,* 8, 690-706.

Tukker, A. 2000. Philosophy of science, policy sciences and the basis of decision support with LCA Based on the toxicity controversy in Sweden and the Netherlands. *The International Journal of Life Cycle Assessment,* 5, 177-186.

Vadenbo, C., Hellweg, S. & Astrup, T. F. 2017. Let's Be Clear(er) about Substitution: A Reporting Framework to Account for Product Displacement in Life Cycle Assessment. *Journal of Industrial Ecology,* 21, 1078-1089.

Velasco-Muñoz, J. F., Aznar-Sánchez, J. A., López-Felices, B. & Román-Sánchez, I. M. 2022. Circular economy in agriculture. An analysis of the state of research based on the life cycle. *Sustainable Production and Consumption,* 34, 257-270.

Vural Gursel, I., Elbersen, B., Meesters, K. P. H. & van Leeuwen, M. 2022. Defining Circular Economy Principles for Biobased Products. *Sustainability,* 14, 12780.

Weidema, B. 2001. Avoiding Co-Product Allocation in Life-Cycle Assessment. *Journal of Industrial Ecology,* 4, 11-33.

Welfle, A., Gilbert, P., Thornley, P. & Stephenson, A. 2017. Generating low-carbon heat from biomass: Life cycle assessment of bioenergy scenarios. *Journal of Cleaner Production,* 149, 448-460.

Wiloso, E. I. 2015. *Development of life cycle assessment for residue-based bioenergy.* PhD thesis, Leiden University.

Yang, Y. 2016. Two sides of the same coin: consequential life cycle assessment based on the attributional framework. *Journal of Cleaner Production,* 127, 274-281.

Yaremova, M., Tarasovych, L., Kravchuk, N. & Kilnitska, O. The evolution of Circular Bioeconomy: A bibliometric review. E3S Web of Conferences, 2021.