Faculty of Agriculture and Forestry

University of Helsinki

Circular bioeconomy in life cycle assessment

- addressing multifunctionality of agriculture

Venla Kyttä

DOCTORAL DISSERTATION

To be presented for public discussion with the permission of the Faculty of Agriculture and Forestry of the University of Helsinki, in Hall 2041, Biocenter 2, on the 25th of November 2022, at 13 o'clock.

Helsinki 2022

Custos
Associate professor Hanna Tuomisto, University of Helsinki

Supervisors

Associate professor Hanna Tuomisto, University of Helsinki

Senior Scientist Merja Saarinen, Natural Resources Institute Finland

Opponent

Associate professor Stefano Cucurachi, Leiden University

Pre-examiners

Senior lecturer Elin Röös, Swedish University of Agricultural Sciences

Associate professor Hannah van Zanten, Wageningen university & research

ISBN 978-951-51-8630-0 (PRINT)

ISBN 978-951-51-8631-7 (ONLINE)

ISSN 2342-5423 (PRINT)

ISSN 2342-5431 (ONLINE)

http://ethesis.helsinki.fi

Dissertationes Schola doctoralis scientiae circumiectalis, alimentariae,

biologicae Universitatis Helsinkiensis

Cover picture: Ernesti Niemi

Unigrafia

Helsinki 2022

Abstract

Due to many serious environmental issues, the production, consumption and disposal of biological resources must shift from a linear system towards a circular bioeconomy. Although the circular bioeconomy has the potential to respond to the changes needed in society, the environmental impacts of various practices must be assessed to identify potential challenges, such as land use competition with food production or negative environmental impacts. Agriculture is a particularly important multifunctional system for the circular economy, which, in addition to its primary function of food production, produces a variety of biomasses that can be utilized as raw material. Furthermore, agricultural systems can recycle by-products and wastes from other industries as fertilizers and soil improvers.

The established method for calculating environmental impacts is the life cycle assessment (LCA) of a product, which aims to capture all impacts caused throughout the entire life cycle of a product system. An allocation procedure is used to partition the environmental impacts of multifunctional systems producing more than one product among all products, resulting in the environmental impact of a single product. This information on the environmental impacts of individual products is typically compared with the impacts of other similar products and used to support decision making. As the allocation method unavoidably affects the results for an individual product, the principles of the allocation methodology must be justified.

This dissertation handles the methodological consideration of the circular bioeconomy in the context of LCA of products from agricultural systems. The objectives of the dissertation are to examine the multifunctionality of agriculture, the different by-products and their recycling, and how the choice of allocation method between products affects the results. Dairy production is used as an example case, since it produces two different foodstuffs as well as various inedible materials, making it an excellent research subject for dealing with multifunctionality.

This dissertation consists of three research papers and a synthesis of them. The first paper (I) examines how the multifunctionality of livestock systems has been addressed

in previous LCA studies through an extensive systematic review of 232 studies. The methodological treatment of recycling and the effect of the choice of method on the results are investigated in paper II through an LCA of recycled fertilizers. The third paper (III) compares the environmental impacts of beef from dairy and beef cattle, pork, cellular meat and tofu production at the system level instead of the level of individual products, considering all the inedible by-products.

The findings of this dissertation demonstrate that, despite attempts to harmonize the LCA method, practices vary between studies and the differences in results due to the choice of method are significant. The relative system-level results also differ significantly from the allocated results for individual products. Hence, allocation should be avoided more often, and comparisons should be made between systems rather than individual products. This would also better reflect the reality, where decision making based on one product inevitably also affects all the other products produced by the system. For example, the environmental impact of milk and beef is usually dealt with separately, even though in reality milk cannot be produced without meat, and decision making on milk alone is not therefore possible. Hence, more system level research is needed.

Tiivistelmä

Useiden vakavien ympäristöongelmien vuoksi uusiutuvien luonnonvarojen tuotannon, kulutuksen ja jätteenkäsittelyn on siirryttävä lineaarisesta järjestelmästä kohti kiertobiotaloutta. Vaikka kiertobiotaloudella voidaan vastata ympäristöongelmien vähentämiseksi tarvittaviin yhteiskunnallisiin muutoksiin, eri toimintatapojen ympäristövaikutuksia on kuitenkin arvioitava optimaalisten tulosten saavuttamiseksi. Maatalous on erityisen tärkeä kiertotalouden monitoimijärjestelmä, joka ruuan lisäksi tuottaa monenlaisia biomassoja, joita voidaan hyödyntää raaka-aineena. Lisäksi maatalousjärjestelmissä voidaan kierrättää muiden teollisuudenalojen sivutuotteita ja jätteitä lannoitteina ja maanparannusaineina.

Elinkaariarviointi (LCA) on vakiintunut tuotteiden ympäristövaikutusten laskentamenetelmä, jonka tavoitteena on ottaa huomioon kaikki tuotejärjestelmän koko elinkaaren aikana aiheutuvat ympäristövaikutukset. Yksittäisen tuotteen ympäristövaikutusten laskemiseksi tuotejärjestelmän ympäristövaikutukset tulee jakaa kaikkien järjestelmän tuottamien tuotteiden kesken allokointimenettelyllä. Usein yksittäisten tuotteiden ympäristövaikutuksia verrataan muiden vastaavien tuotteiden vaikutuksiin ja vertailujen tuloksia käytetään päätöksenteon tukena. Koska laskennassa käytetty allokointimenetelmä vaikuttaa väistämättä yksittäiselle tuotteelle laskettaviin ympäristövaikutuksiin, tulee käytetyn allokointimenetelmän olla perusteltu.

Tässä väitöskirjassa tutkitaan kiertobiotalouden metodologista käsittelyä maataloustuotteiden elinkaariarvioinnissa. Väitöskirjan tavoitteena on tarkastella maatalouden monitoiminnallisuutta, erilaisia sivutuotteita ja niiden kierrätystä sekä sitä, miten tuotteiden välinen allokointimenetelmä vaikuttaa tuloksiin. Esimerkkitapauksena käytetään maidon tuotantoketjua, sillä se tuottaa kahta erilaista elintarviketta sekä erilaisia syötäväksi kelpaamattomia materiaaleja, mikä tekee siitä erinomaisen tutkimuskohteen monitoiminnallisuuden käsittelemiseksi.

Väitöskirja koostuu kolmesta tutkimusartikkelista ja niiden yhteenvedosta. Ensimmäisessä tutkimuksessa tarkastellaan laajan 232 tutkimuksen systemaattisen kirjallisuustarkastelun kautta, kuinka maidon ja naudanlihan tuotannon

monitoiminnallisuutta on käsitelty aikaisemmissa LCA-tutkimuksissa. Toisessa artikkelissa tutkitaan kierrätyksen metodologista käsittelyä ja menetelmän valinnan vaikutusta tuloksiin kierrätyslannoitteiden elinkaariarvioinnin avulla. Kolmannessa tutkimuksessa verrataan naudanlihan (lypsy- ja lihakarjasta), sianlihan, viljellyn lihan ja tofun tuotannon ympäristövaikutuksia järjestelmätasolla yksittäisten tuotteiden vertailun sijaan, huomioiden myös kaikki syötäväksi kelpaamattomat sivutuotteet.

Tämän väitöskirjan tulokset osoittavat, että huolimatta LCA-menetelmän harmonisointiyrityksistä, käytännöt vaihtelevat tutkimusten välillä ja menetelmän valinnasta johtuvat erot tuloksissa ovat merkittäviä. Suhteelliset järjestelmätason tulokset eroavat myös merkittävästi yksittäisille tuotteille allokoiduista tuloksista. Järjestelmätason vertailu kuvastaa myös paremmin todellisuutta, jossa yhteen tuotteeseen perustuva päätöksenteko vaikuttaa väistämättä myös kaikkiin muihin saman järjestelmän tuottamiin tuotteisiin. Esimerkiksi maidon ja lypsykarjasta peräisin olevan naudanlihan ympäristövaikutuksia käsitellään yleensä erillisinä, vaikka todellisuudessa päätöksentekoa ei voida kohdentaa vain yhteen tuotteeseen. Näin ollen lisätutkimusta tarvitaan järjestelmätason vaikutusten arvioimiseksi.

Acknowledgements

First, I want to express my deepest gratitude to my supervisors, Associate Professor Hanna Tuomisto from the University of Helsinki and Senior Scientist Dr Merja Saarinen from Natural Resources Institute Finland, for all the encouragement, support and help I have received during this process. I have learned a lot from both of you. I would also like to thank my co-authors and all my colleagues for their contributions and insights during these years. I am grateful for all the inspiring and helpful conversations that I have had with you. Warmest thanks to my co-author, Dr Marja Roitto, for always listening to my worries and offering help, and PhD Candidate Lotta Kaila for your friendship and peer support during this project.

I want to thank Associate Professor Stefano Cucurachi from Leiden University for being my opponent in the public examination of this thesis, as well as Dr Elin Röös from the Swedish University of Agricultural Sciences and Associate Professor Hannah van Zanten from Wageningen University & Research for acting as the preliminary examiners. I also want to acknowledge the financial support received from the Finnish Association of Academic Agronomists, the August Johannes and Aino Tiura Agricultural Research Foundation, and the Foundation for Nutrition Research.

This work was conducted during 2020–2022, which means that most of it was completed at home in isolation due to the pandemic. Therefore, I also want to thank my home office assistants, Viki and Haddock, for keeping me company every day.

During this project, I have received enormous support from friends and family, for which I am deeply grateful to each and every one. I especially want to thank my parents, Tarja and Olavi, for encouraging me to study and supporting me through all these years, and my sister Kaisa for always having my back and cheering me up. I also want to thank Dr Marko Lamminsalo, for all the help and peer support I have received. Lastly, I want to thank Ernesti, without whom I would not have been able to get through this project in the first place. You are all very dear to me.

Table of Contents

List of original publications

This thesis is based on the following publications, later referred as paper I, II & III:

I. Kyttä, V., Roitto, M., Astaptsev, A., Saarinen, M. & Tuomisto, H. L. (2022). Review and expert survey of allocation methods used in life cycle assessment of milk and beef. The International Journal of Life Cycle Assessment 27, 191–204. https://doi.org/10.1007/s11367-021-02019-4

II. Kyttä, V., Helenius, J., & Tuomisto, H. L. (2021). Carbon footprint and energy use of recycled fertilizers in arable farming. Journal of Cleaner Production, 287, 125063. https://doi.org/10.1016/j.jclepro.2020.125063

III. Kyttä, V., Roitto, M., Saarinen, M. & Tuomisto, H. L. (unpublished). Comparison of individual food products may underestimate the underlying environmental impacts.

The contribution of the authors in the original articles of the thesis:

Paper I:

V.K.: Methodology, Formal analysis, Investigation, Writing - Original Draft, Writing - Review & Editing, Visualization **M.R.**: Conceptualization, Writing - Review & Editing, Funding acquisition **A.A.**: Conceptualization, Writing - Review & Editing **M.S.**: Writing - Review & Editing **H.T.**: Conceptualization, Writing - Review & Editing, Supervision, Funding acquisition

Paper II:

V.K.: Conceptualization, Methodology, Investigation, Writing - original draft, Writing - review & editing, Visualization. **J.H.**: Conceptualization, Writing - original draft, Supervision. **H.T.**: Conceptualization, Methodology, Writing - original draft, Writing - review & editing, Supervision

Paper III:

V.K.: Methodology, Formal analysis, Investigation, Writing - Original Draft, Writing - Review & Editing, Visualization **M.R., M.S.**: Writing - Review & Editing **H.T.**: Methodology, Writing - Review & Editing, Supervision

Abbreviations

Apos – at the point of substitution

AS – ammonium sulphate

BD – biogas digestate

CFF – circular footprint formula

FU – Functional unit

GHG – greenhouse gas

LCA – Life cycle assessment

MBM – meat and bone meal

MF – mineral fertiliser

N – nitrogen

P – phosphorus

1. Introduction

1.1 The role of agricultural systems in the circular economy

Human activities place significant strain on the Earth's environmental boundaries, endangering ecosystem stability and biodiversity by causing significant changes in atmospheric conditions, nutrient flows and land systems (Rockström et al., 2009; Steffen et al., 2015). Agriculture is a significant or major driver of the crossing of the planetary boundary for the safe operation of the Earth (Campbell et al., 2017; Poore & Nemecek, 2018).

Due to multiple and severe environmental issues, the need for changes in biological resource production, consumption, processing, storage, recycling and disposal is recognized, and the development of a circular bioeconomy has been proposed as a solution (European Commission, 2012; Hetemäki et al., 2017). The circular bioeconomy combines the two distinct concepts of circular economy and bioeconomy with the goal of developing a strategy based on regenerative and restorative systems that produce renewable materials that are used to produce products that maintain their highest possible utility and value at all times. Therefore, agriculture, as a producer of biomass, has an essential role in the circular bioeconomy strategy (Fig. 1).

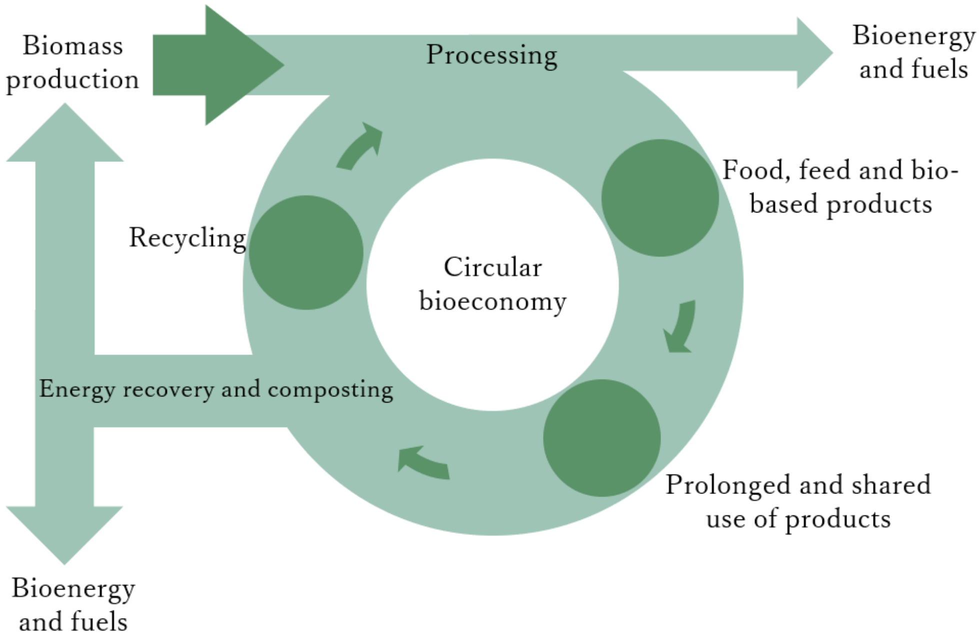

Fig. 1. *The circular bioeconomy concept. Altered from (Stegmann et al., 2020).*

From a linear life-cycle perspective, typical industrial agriculture demands inputs (such as agrochemicals and machinery) and produces food and other materials, e.g., fibres, as outputs. In a circular bioeconomy, the whole system is based on agriculture and forestry as the producers of biomasses to be utilized as food, feed, bio-based products and energy sources. In such a system, the residues are also considered to be important feedstocks (Stegmann et al., 2020). At the same time, agricultural systems can use residues and wastes from other industries as fertilizers and soil amendments, effectively recycling them back to biomass (Huygens & Saveyn, 2018; Möller et al., 2018).

Even though there are high hopes for the circular bioeconomy to meet the changes required for society to remain within planetary boundaries, it is not sustainable by default and must be designed as such (Hetemäki et al., 2017; Saidani et al., 2022; Stegmann et al., 2020), because products with high circularity do not necessarily have low environmental impacts (Roos Lindgreen et al., 2021). Therefore, it is critical to identify potential trade-offs, such as competition with food production or negative environmental impacts. Information on the environmental impacts of different products and services is increasingly desired. To define the environmental impacts arising from different sources and to target actions accordingly, appropriate assessment methods for impacts are crucially needed. Since comparisons are usually made between products providing the same function, the underlying allocation of impacts must be founded on sound and justified methodology.

1.2 Life cycle assessment of agricultural products

The current established method for product-level analyses is life cycle assessment (LCA), which aims to capture environmental impacts caused during the whole life cycle of a product or service (ISO 14040:2006, 2006). The LCA methodology was originally designed for the linear economy, covering the life cycle from "cradle to grave". To assess the products of the circular bioeconomy, the method needs to be applicable and appropriate to also catch the impacts of complex non-linear life cycles.

LCA is widely applied in estimating the environmental impacts of agricultural products. By default, the assessment procedure produces product-specific results (ISO 14040:2006, 2006), which are often also used to compare different products with each other. The multifunctionality of agricultural systems is usually handled by using allocation to partition impacts between the main product and by-products. As the majority of the environmental consequences of food occur at the farm level, the allocation choices used in agricultural product assessments can have a significant impact on the outcomes of LCAs of food products, and also on the outcomes of LCAs of products based on agricultural by-products, whose role is growing as the circular bioeconomy grows. The debate over how allocation should be done and on what the method should be based on has been going on for decades, but no method suitable for all situations has been found (Christel Cederberg & Stadig, 2003; Chen et al., 2017; Ijassi et al., 2021; Tillman et al., 1994; Wilfart et al., 2021).

1.2.1 Attributional and consequential approaches

Two different approaches for LCA have been adapted: attributional LCA and consequential LCA. Attributional models include the processes that have contributed to the life cycle of a product, "tracing the contributing activities backward in time" (Weidema, 2014). Consequential models, in turn, include the activities that are predicted to change when the product under study is produced, consumed and disposed (Weidema, 2014). Since the LCA ISO standard does not differentiate between these two approaches, combinations of these two are sometimes also used.

In attributional modelling, multifunctionality is commonly solved through allocation, leading to separate results being obtained for each product. In consequential modelling, by-products are handled by substitution, meaning that the provisioning of a by-product is subtracted from the processes, also leading to a result for a single product.

1.2.2 Categorizing the outputs of a system in LCA

The ISO standard defines a product as any good or service (e.g., processed or unprocessed material) and by-products as "two or more products coming from the same unit process or product system" (ISO 14040:2006). Waste is defined as "substances or objects which the holder intends or is required to dispose of." According to these definitions, all output material flows not going to disposal are to be handled as products in LCA.

In LCA studies, the products are often defined based on whether they have economic value (European Commission, 2010). As a result, materials with zero impacts occur, while other products may be over-burdened. This may lead to bias favourable to systems using these materials as raw material. The question of defining and handling materials in LCA is highlighted as the world transits towards a circular economy. Since one of the fundamental goals of the circular economy is to reduce the amount of waste generated and promote the usage of waste and devalued materials, the shift towards a circular bioeconomy will redefine the current view on materials.

1.2.3 Allocation to outputs in LCA

Although the LCA method is standardised, different interpretations and practices exist, increasing the uncertainty of the method and complicating the comparison of results from different assessments. One critical methodological choice is how the output materials of a system are classified into products (functions) or wastes, and how the impacts are assigned between these outputs. The basic principle is that all impacts arising from a process, including waste treatment, are allocated between products (Fig. 2) (European Commission, 2010; ISO 14040:2006, 2006). In addition to classification into products or wastes, materials are sometimes also classified as residues, meaning materials that are utilized but have no economic value and no upstream impacts allocated to them (e.g., manure) (EDA, 2018).

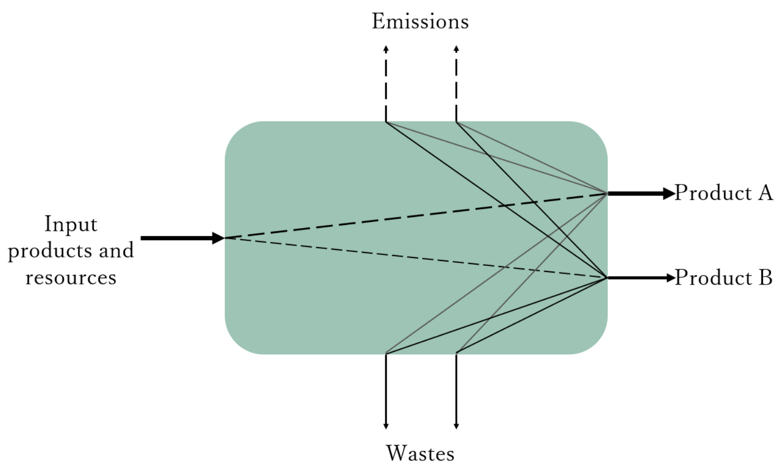

Fig. 2. Handling the multifunctionality of a process/system (production of products A & B) by allocating the impacts of inputs, emissions from the process and waste handling between the output products. Adapted from (European Commission, 2010).

The allocation method chosen needs to be applicable to the systems under study, ensuring that the impacts are assigned to products appropriately. Since LCA is a comparative method, methodological consistency needs to be achieved in order to ensure the comparability of different products (Tillman et al., 1994). The LCA standard (ISO 14040:2006, 2006) guides solving the multifunctionality issues by following the given allocation procedure:

1. Allocation should be avoided either by separating unit processes into sub-processes and assigning inputs and outputs correspondingly or by including by-product functions in the assessment through system expansion.
2. Allocation of the impacts of the inputs and outputs following the physical relationships between them, reflecting the material balances between inputs and outputs.
3. Allocation following other than physical relationships, for example allocation based on the economic values of the products.

1.2.4 Different allocation approaches

Allocation can be avoided through system expansion, which is based on a concept introduced by Tillman et al. (1994). In this 'technological whole system' approach, the demand for functions provided by the system is assumed to be stable, and all the products that are affected by changes in demand are consequently included inside the system boundaries of an assessment. This needs to be considered when comparing different systems, and compared systems are therefore complemented to include similar functions, such as equivalent products or services fulfilling the same purposes. In practice, this is ensured by adding or subtracting alternative products providing the same functions as the by-products originating from one of the compared systems, until all the systems include the provision of the same functions (Fig. 3).

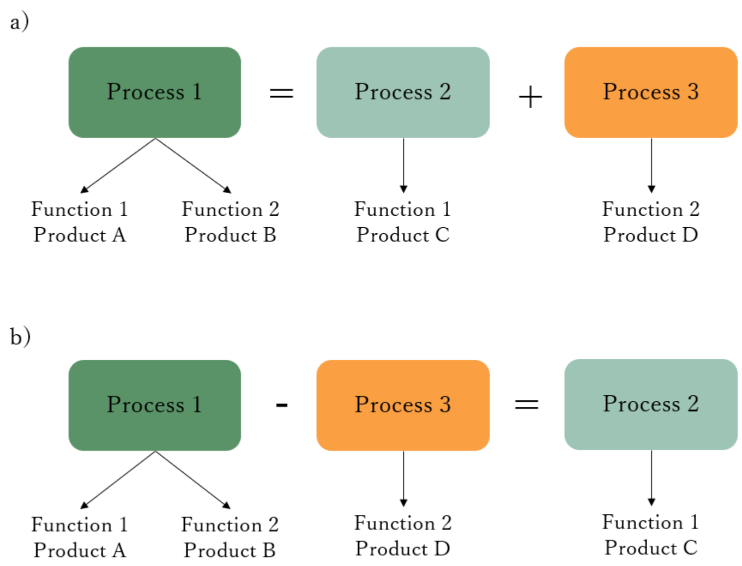

Fig. 3. *Principles of adding (a) or subtracting (b) processes to make compared systems fulfil all the same functions. Adapted from (Tillman et al., 1994).*

Even though Tillman et al. (1994) presented adding and subtracting processes as equivalent methods for making different systems comparable, subtracting has since been separated as a method for consequential assessments (Weidema, 2003). Probably due to the origin of the method, system expansion and substitution are often interpreted to be synonymous (Heijungs et al., 2021). In this thesis, the term system expansion is used to mean a procedure for complementing a system by adding processes to it, and substitution refers to subtracting processes from a system.

Biophysical allocation is based on physical causality, reflecting how the inputs of a process are utilized by each output. It is able to catch the underlying physical relationship and is therefore preferred in the ISO allocation procedure over other allocation methods. There is no established biophysical allocation method for all multiproduct systems, but for milk, for example, biophysical allocation is based on the mathematical relationship of feed energy converted to tissue or milk (Thoma et al., 2013, updated by Ineichen et al., 2022).

Economic allocation is a widely applied allocation method that uses the prices and volumes of outputs as allocation criteria. Therefore, economic allocation also classifies output materials as products or wastes based on whether they have economic value or not. Instead of physical realism, economic allocation reflects the socioeconomic cause of impacts, indicating the value of different products to society (Pelletier et al., 2015).

Mass allocation is solely based on the masses of outputs, and environmental impacts are allocated accordingly. Allocation can also be based on other relationships between the materials, such as dry matter mass, protein, energy, nitrogen or fat.

1.2.5 Recycling in LCA: allocation between systems providing and systems using recycled materials

The core of the circular economy is keeping products and raw materials in use for as long as possible, and it is therefore important to have appropriate methods to handle recycling in LCA. Generally, the reuse and recycling of materials can be divided into two classes: closed-loop systems and open-loop systems (ISO 14040:2006, 2006). In closed-loop systems, there is no change in the properties of the material, and secondary material can be directly used to replace virgin material in the same product system. In open-loop systems, the material undergoes changes in its properties and is recycled to be used in other product systems. The general allocation principles also apply to the recycling of materials (ISO 14040:2006, 2006). However, in practice, many LCA guidelines have separate recommendations for handling by-products and recycled materials, even though the categorization of materials into these classes is often unclear (Schrijvers et al., 2016).

When a material is processed into different products or used in other systems for other purposes, the impacts need to be allocated between the systems providing the material and systems using the material. Different methods have been developed to assess recycling between systems and the assessment of credits from the displacement of other materials (Allacker et al., 2014). For example, the European Commission has developed the Circular Footprint Formula (CFF) to allocate the impacts arising from the recycling process and transportations between the systems providing and the systems using the material (European Commission, 2018). In addition, substitution is commonly used, even in attributional models. For instance, the fertilizer use of organic residues is often assessed by defining the quantity of mineral fertilizers replaced (Brockmann et al., 2018; Hanserud et al., 2018; Spångberg et al., 2011).

Often, materials that are going to be recycled have no economic value at the point where they are separated but are valuable materials after the recycling process. Therefore, the recycling process can be included in the process where the recyclable materials are formatted, and allocation can then be carried out between the main products and the recycled products. This approach is called allocation at the point of substitution (APOS) and is used, for example, in ecoinvent databases (Wernet et al., 2016). Another methodological approach used in the ecoinvent database is known as cut-off, where the impacts of recyclable materials are "cut" at the start of the treatment operations, making them available burden-free for further uses. Following this method, waste treatment is fully allocated to the producer of the waste and all valuable materials obtained from waste treatment are available burden-free. However, the main agricultural product, food, differs significantly from other materials and products: the life cycle of food products is relatively short, they can be consumed only once, and the characteristics of the products change during the consumption phase. Therefore, the recycling loop of food is in practice the recycling of nutrients.

1.2.6 Impact of allocation methods

In a circular economy, the recycling of materials and utilization of by-products increases. As a consequence, the life cycles of products become longer and more complex, and hence also the allocation choices, and the uncertainty related to allocation multiplies. The impact of allocation methods, especially in the case of animal products, has been widely investigated, showing displaying notable method-dependent variation in the results (Cederberg & Stadig, 2003; Chen et al., 2017; Flysjö et al., 2011; Gilardino et al., 2020; Kiefer et al., 2015). As the allocation method affects the results, efforts have been made to harmonize the LCA method to ensure the comparability of different assessments (European Commission, 2013, 2018).

2. Objectives of the thesis

The overall objective of this thesis is to **examine the methods used for handling by-products in the life cycle assessment of products from agricultural systems** in the circular bioeconomy. The more specific objectives are to investigate the following:

1) How and on what basis are different outputs of systems classified as products or wastes?

2) How are environmental impacts allocated between different outputs and how does this affect the results?

3) How are differences in the quality and usage of materials considered?

The research uses dairy production as example case, since it produces two different foodstuffs and various inedible materials, hence being an excellent subject of research for handling multifunctionality on the product level. The objectives of the thesis are met through three studies that:

1) Provide information on how different allocation methods and the inclusion of by-products affects the environmental impacts of milk and beef from dairy production and clarify the rationale behind the use of different allocation methods;

2) Assessed the impacts of different allocation methods between systems providing and using recycled materials in the case of recycled fertilizers (such as biogas digestate and meat and bone meal);

3) Developed and tested a method to avoid allocation between milk and meat and inedible by-products, such as hides, bones, fat, internal organs and blood.

The publications based on these studies (papers I–III) address the usage of and justification for different allocation methods from aspects of defining outputs as products or wastes, the handling recycling/circulation of materials and making products comparable.

A summary of the issues addressed, the methods and the outcomes of each paper (I–III) is presented in Table 1. Even though the multifunctionality of agriculture also undeniably has immaterial, social and economic dimensions, this thesis focuses on physical materials and environmental aspects on the product level.

Table 1. *Summary of the papers responding to the research objectives.*

Paper	*Issues handled*	*Method*	*Outcome*
I	Classification of products and wastes, justification of different methods, the dependency on the study goal	Structured literature review of allocation methods and justifications for using certain methods	Review of the state of the art and aspects affecting allocation choices
II	Impact of the classification of outputs and consequent methods of handling recycling	LCA on the usage of recycled materials as fertilisers	Testing of methods to handle the recycling and circulation of materials in LCA
III	Inclusion of all functions derived from different production systems, making different systems comparable with each other	Avoiding allocation through LCA of whole production systems	Development and testing of a method to include the multifunctionality of systems in LCA

3. Materials and methods

The issues discussed in earlier chapters regarding the multifunctionality of agricultural systems were examined through a structured literature review and two case studies, each described more closely in their own sections below. A detailed report on the materials and methods used is presented in the corresponding original publications (papers I–III).

3.1 Review of allocation methods and LCA practitioners' perceptions (Paper I)

A structured literature review was conducted to analyse the different allocation methods used in milk, beef meat and inedible by-product studies and to explore the rationale of LCA practitioners for using specific allocation methods. First, all the milk and beef environmental impact studies, both peer-reviewed and non-peer-reviewed, collected by Poore & Nemecek (2018) were reviewed, and after that, further literature searches were conducted, first in Google Scholar and later in Scopus (Fig. 4). To be included in the review, the studies were required to be published in 2000 or thereafter, accessible, use LCA methodology, report the allocation method used, calculate at least the carbon footprint and report the results in a numerical form per output unit (e.g., FPCM, live weight).

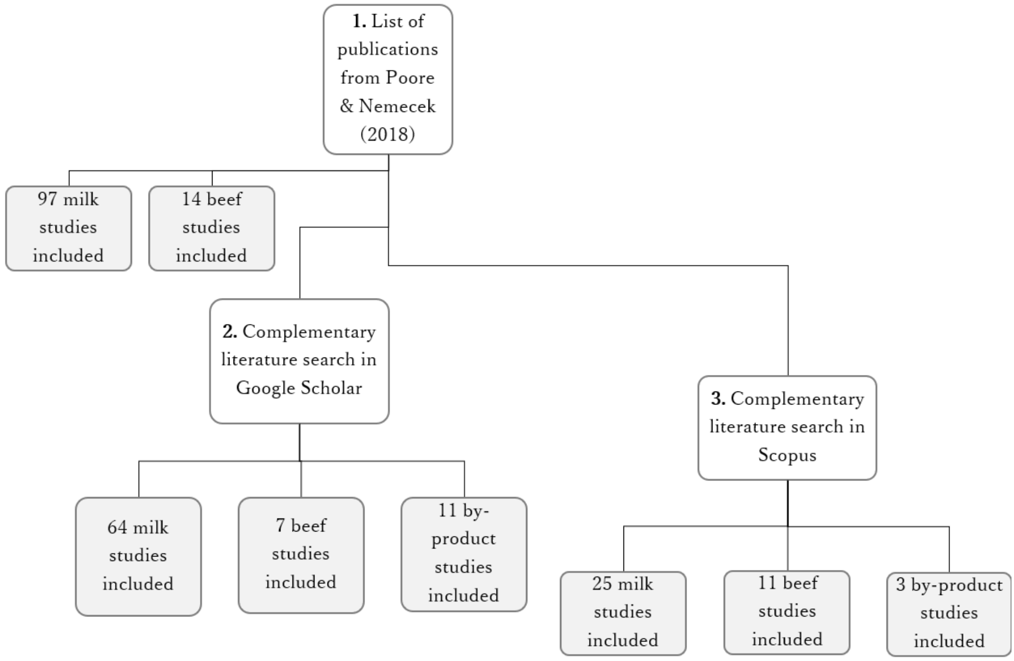

Fig. 4. *Workflow of the literature search and the resulting number of included studies.*

The following information was gathered from the included studies: the product being researched, the publication type, the purpose of the study, the system boundary, the functional unit and the allocation method used in the study. To evaluate the impact of the study goal on the allocation method, the reported goals of the studies were classified as follows: comparative studies (comparing different production systems or scenarios of the same product), comparative studies comparing different products (e.g., beef with other meats), studies quantifying the environmental impacts of a product (typically seeking hotspots), methodological studies developing, testing or comparing methods or methodological approaches, consequential LCA studies, and studies that do not fit into any of the above categories.

To survey the argumentation behind the selection of a particular allocation method, a survey was sent to the authors of the studies included in the review, and the reasonings presented in the studies were collected. The survey further mapped out the opinions of LCA practitioners regarding the strengths and weaknesses of different allocation methods, the classification of outputs into products, residues and wastes, and the appropriate methods for allocation between different outputs.

3.2 Recycling of materials: A case study (Paper II)

The aims of this study were to assess the GHG emissions of recycled fertilizers, compare the different recycled fertilizer and mineral fertilizers, and test and compare different allocation methods for the recycled fertilizers. LCA was used to estimate the GHG emissions of oat (*Avena sativa*) production when using the recycled fertilizers ammonium sulphate (AS), biogas digestate (BD) and meat and bone meal (MBM). As a comparison, mineral fertilizers (MF) with two different application rates (107 and 160 kg N ha^{-1}) and no fertilization were also included. Yield trials for the spring sown oat cultivar "Obelix", carried out as field experiments in southern Finland during the growing season 2017, were used as a basis for the assessment. GHG emissions were modelled with OpenLCA 1.10.2 software along with the embedded ecoinvent 3.6 (2019) database (openLCA; Wernet et al., 2016).

AS can be produced from a variety of raw materials such as side streams of the nickel industry and from livestock manure, but in this study, it was assumed to originate from nylon production. The MBM used in fertilizers is produced from animal by-products such as carcasses collected from farms or rejected material from slaughterhouses. The components of the fertilizer consist of meat and bone meal, oat hulls, vinasse and granulated poultry manure. For biogas production, several agricultural side streams, such as manure and plant waste, as well as municipal wastes such as biowaste and wastewater, are suitable raw materials (Möller and Müller, 2012). In this study, biogas was assumed to originate from grass.

For the functional unit of tons of oats, the baseline system boundary included the manufacturing of fertilizer for AS, MBM and MF, fertilizer application for AS and BD, and transportation, sowing, harvesting, direct soil emissions, and indirect soil emissions for all fertilizers. For the functional unit of N kg, only manufacturing, transportation, fertilizer application and soil emissions were included. The changes in soil carbon were not assessed.

In the baseline system model, the materials of recycled fertilizers were considered to be residues, and no burdens or credits were allocated to them. The processing of animal by-products into MBM and fat was allocated between these products based on mass (62% to MBM).

In some situations, the raw materials can be classified as products or wastes. Some recycled fertilisers are on markets, and hence also have a price. According to the ILCD handbook (European Commission, 2010), the economic value can be used as indication of the material being a product, and a scenario where the raw materials were classified as products with impacts allocated to them was therefore created. Then again, the raw materials of fertilisers can be wastes with no or negative economic value for the systems that produce the material. Some animal by-products, for example, have a negative economic value for slaughterhouses, since they must pay for the treatment of the materials.

In order to compare the impacts of allocation methods on the results, handling raw materials as products with economic allocation or as wastes with the Circular Footprint Formula (CFF) developed by the European Commission's Environmental Footprint project (European Commission, 2018) was tested for BD and MBM. A summary of the tested approaches is presented in Table 2.

Table 2. Classification of materials and consequent allocation approach used. CFF = Circular Footprint Formula

Classification of material	Residue	Product	Waste
Classification basis	A material that is utilized (hence no waste) but has no economic value	Is used to replace mineral fertilizers and hence has economic value	A material that is intended for disposal, with no or negative economic value
Allocation method	No allocation	Economic allocation	Impacts arising from the recycling process allocated between systems by using the CFF

Following the ISO 14040:2006 allocation procedure, economic allocation was applied when raw materials were handled as products, since it can be consistently applied in every stage of the life cycle. Allocation based on a physical relationship (e.g., mass) was not applied, since it was considered as an unequal approach for different uses of the materials.

The economic allocation scenario for BD was modelled based on ecoinvent data concerning biogas production from grass. The feedstock was assumed to have the same nutrient content as the digestate in the baseline scenario. Economic allocation was calculated based on the N content of digestate, resulting in 4.3% of impacts being allocated to digestate.

MBM was assumed to be half of cattle and half of swine origin. The proportion of nonedible by-products was set to 30% for swine and 55% for cattle (Gac et al., 2014). Economic allocation for cattle category 3 slaughter by-products (0.8%), suggested by the Cattle Model Working Group, was applied for both cattle and swine by-products (JRC, 2016). Impacts from oat production to oat hulls were assessed based on the mass and economic allocation (0.44%) information for the oat mix fraction (Heusala et al., 2020). For vinasse and dried and pelletized poultry manure, data and prices from ecoinvent were used.

When the raw materials were classified as wastes, CFF was used to allocate the impacts arising from the waste treatment process. CFF handles the materials as recyclable waste from other systems and thus the materials are not burdened with impacts from the system providing the material, but impacts from the recycling process and transportation linked to the recycling process are allocated between the systems providing and the systems using the material. The processes for which the CFF was applied were anaerobic digestion for BD and rendering, poultry manure drying and pelletizing, vinasse processing and transportation of these materials for MBM. Oat hulls are not processed before use in fertilizer manufacturing, and the recycling process for oat hulls was not therefore included.

The CFF formula consists of three parts: material, energy and disposal. Since this study was a cradle-to-gate instead of a cradle-to-grave study, energy and disposal formulas were equal to 0 (European Commission, 2018). Therefore, only the formula for material was used. In cradle-to-gate studies, two allocation factors (A) of burdens and credits between the supplier and user of recycled materials are used, A = 1 as a default (all burdens to the user) and material-specific A = 0.5 (European Commission, 2018). The point of substitution was set to after raw-material processing, before MBM pelletizing and BD transportation to the farm. Between the output products of processing, the same economic allocation factors as in the economic allocation scenario were used.

3.3 Avoiding allocation through expanded system boundaries: A case study and methodological framework (Paper III)

The aim of this study was to compare whole production systems instead of single food products, using the dairy cattle system as a baseline. The dairy cattle system was compared with beef cattle, pork production, cultured meat production and tofu production. Instead of allocation, the product systems were made comparable by using system expansion through adding functions to the compared systems.

The functional unit was standardised to include similar functions by adding alternative products that could serve as substitutes for the by-products of the dairy system. The products derived from the dairy system were milk, food for human consumption (meat and edible organs), leather, pet food and feed, meat and bone meal as a fertiliser, biodiesel, biogas, and biogas digestate as a fertiliser. In addition, lactic acid, which is formed as a by-product in cultured meat production, was included in the assessment and added to the compared systems.

Products were assessed to the point where they are substitutable (ready for use, fulfilling the same functions), and the use phase and end-of-life treatment were not therefore included. To make the compared systems equivalent, average market consumption mix datasets were used as alternative products. If the market dataset of a product could not be defined or did not exist, the most common or most likely alternative single product with similar properties was used. The compared systems with added alternative products are presented in Table 3.

System expansion was only applied to foreground processes (production of the main products), since including the multifunctionality of every background process in every system would lead to endlessly growing systems. The background processes (by-product processing and production of inputs) were included as they are structured in database

Table 3. *Compared systems of dairy cattle, beef cattle, pork, cultured meat and tofu, including the products produced by each system and the products added to each system to make the functional unit equivalent. The amounts of pet food and animal feed alternatives differ from each other due to their different protein content.*

Product/ Main system	Meat/tofu, kg	Milk[a], kg	Leather[b], kg	Pet food/ feed[c], kg	MBM[d], N g	MBM[e], P g	Biodiesel[f], kg	Biogas[g], m^3	Digestate[d], N g	Digestate[e], P g	Lactic acid[h],kg
Dairy cattle	1.00	70	0.02	0.54	6	4	0.052	0.033	1.2	0.2	
Added products											0.77
Beef cattle	1.00		0.02	0.54	6	4	0.052	0.033	1.2	0.2	
Added products		70									0.77
Pork	1.00			0.13	4	3	0.034	0.021	0.8	0.1	
Added products		70	0.02	0.51	2	1	0.018	0.012	0.4	0.1	0.77
Cultured meat	1.00										0.77
Added products		70	0.02	0.67	6	4	0.052	0.033	1.2	0.2	
Tofu	1.00			0.34							
Added products		70	0.02	0.33	6	4	0.052	0.033	1.2	0.2	0.77

Alternative product added: [a] market for soybean beverage (ecoinvent), [b] market for PVC (ecoinvent), [c] market dataset for pet food (created), [d]market for N fertiliser (ecoinvent), [e] market for P fertiliser (ecoinvent), [f] market for diesel (ecoinvent), [g] market for natural gas (ecoinvent), [h] market for lactic acid (ecoinvent)

Data concerning the masses and uses of beef by-products were derived directly from a meat company and a company processing the animal by-products, and additional data were obtained from the literature and databases. The systems were modelled with the ecoinvent (APOS) 3.6 database (Wernet et al., 2016) in OpenLCA 1.10.2 software by using the ReCiPe 2016 Midpoint (H) (V1.1) impact assessment method (Huijbregts et al., 2017). Ecoinvent market activity data with global coverage were used to model all the processes.

To test how the results from comparing allocated single products differ from the comparison of system-level results, the impacts of meat (and edible offal) originating from dairy cattle, beef cattle and pork, cultured meat and tofu were assessed using economic allocation, including impacts from slaughtering. For bovine and pork meat, the default economic allocation factors defined by FEDIAF (2018) were used, being 92.9% for beef and 98.9% for pork. The estimated price of cultured meat varies greatly depending on the production technology (CE Delft, 2021; Humbird, 2020), but in this study, the estimated affordability threshold of $25/kg cultured meat in the early market stage was used (Humbird, 2020). The economic value of lactic acid was obtained from the ecoinvent database, resulting in 95% of impacts being allocated to cultured meat. For tofu, ecoinvent data were used, resulting in 79% of impacts being allocated to tofu.

4. Results

4.1 Allocation methods used between products (Paper I)

The evaluation of allocation strategies revealed notable differences in methodological approaches. Biophysical allocation and economic allocation were found to be the most generally utilized methods in LCAs of milk and beef, whereas protein, mass, energy and system expansion were mainly used in studies comparing different methods, or in sensitivity analyses of studies to indicate the impact of an allocation method (Fig. 5). Only a few studies included inedible products and allocated impacts to them. Some studies also reported allocating all impacts to a single product because allocation had no relevance for the study goal (e.g., comparing the impact of mitigation practices).

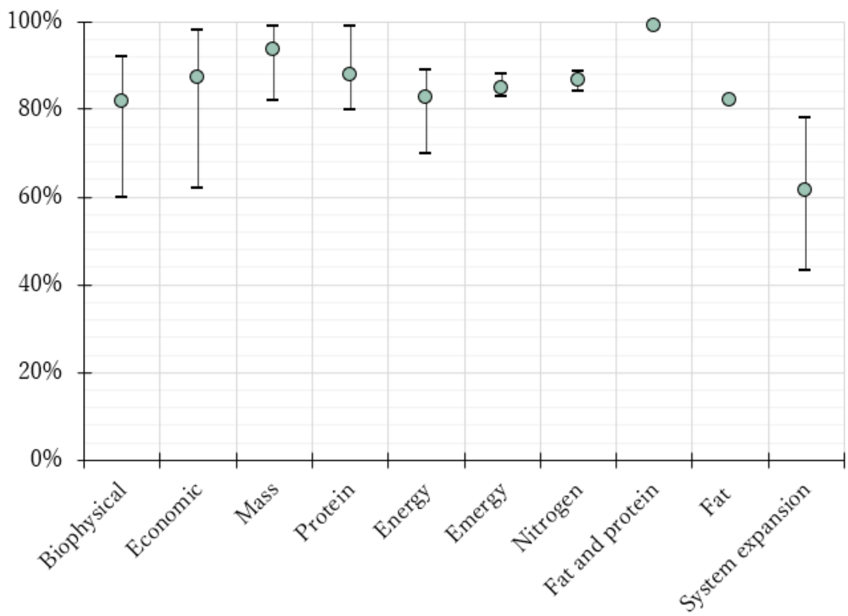

Fig. 5. Average, minimum and maximum share of impacts allocated to milk in LCA studies applying different methods.

The allocation factors resulting from different methods differed significantly between beef LCA studies and LCA analyses of products containing beef by-products (Fig. 6). Economic, mass and energy allocation were the most frequently used. Allocating according to the energy content or biophysical allocation resulted in a higher share of impacts being assigned to by-products (52% and 73%) than to meat. In addition, mass allocation resulted in a relatively low share of impacts being allocated to meat (55%) compared to economic allocation (87% to meat).

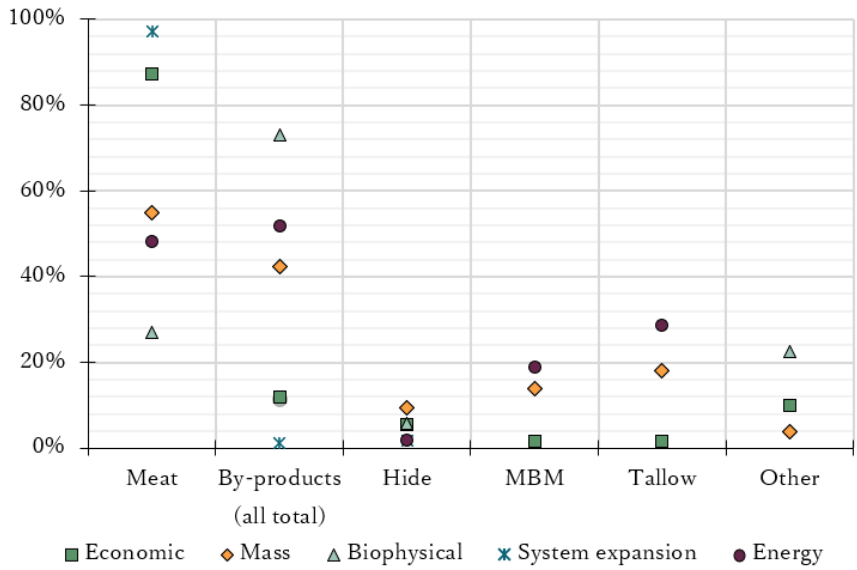

Fig. 6. Average allocation ratios used in LCA studies of beef and products utilizing beef by-products as the raw material. Meat also includes other edible parts. The category of 'by-products (all total)' represents the combined total allocated to all by-products, and subsequent categories represent a more detailed breakdown between by-products if more detailed data were available.

Although system expansion is the first allocation step in the ISO standard allocation procedure, it is less frequently used and is usually only included for comparison of different allocation methods. In the studies, system expansion was only implemented as substitution, i.e., by assessing impacts from avoided production.

Classification according to study goals demonstrated that the share of different goals was similar, regardless of the allocation method, except that method research was more common in studies using several allocation methods and system expansion was more common in studies comparing different systems (Fig. 7). Biophysical allocation was not used in a single study aiming to compare different products.

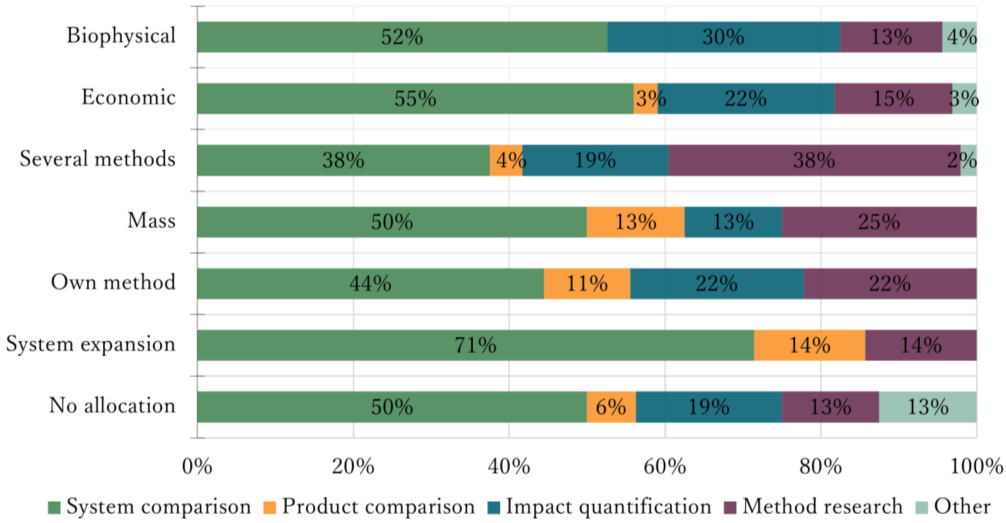

Fig. 7. *Study goals in studies using a particular allocation method. Details concerning the classification are presented in the Materials and methods section. Methods that were used in less than five studies were excluded from the figure.*

The majority of the studies presented reasoning to support the methodological choices of allocation between by-products. The most common reasonings were that the method was recommended in a particular LCA guideline (especially for biophysical allocation), and that the method had been used in other similar studies. The general argumentation for using economic allocation was weak, but in the survey, most of the LCA practitioners still replied that they would use economic allocation for all products: milk, meat, manure and inedible body parts.

4.2 Allocation between systems (Paper II)

In the baseline scenario, the climate change impacts of mineral fertilizers were higher than recycled fertilizers for both the functional unit of yield ton and kg of N (Fig. 8). The biggest differences between fertilizers occurred in manufacturing, whereas impacts from other processes were quite similar. For BD, there were no impacts from manufacturing in the baseline scenario, but the share of transportation was substantially higher.

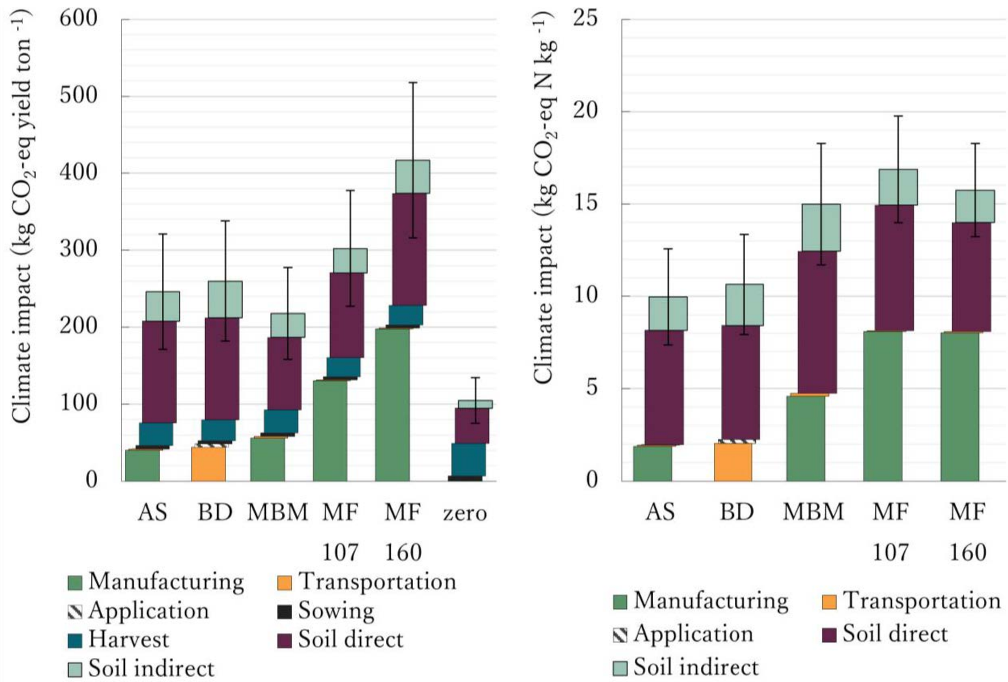

Fig. 8. *Baseline climate impact (kg CO_2-eq) of fertilizers per yield ton and kg of fertilizer N. The error bars represent the standard error based on Monte Carlo analysis. AS = ammonium sulphate; BD = biogas digestate; MBM = meat and bone meal; MF = mineral fertilizer (N ha^{-1}); zero = no fertilization.*

The method for handling the raw materials of recycled fertilizers (residue, waste, by-product) affects the system boundaries and the burdens of the raw materials (Fig. 9). Handling the materials as products results in impacts from primary production being allocated to fertilizers and thus in higher impacts. Applying economic allocation instead of the mass allocation used in the baseline model for MBM resulted in smaller GHG emissions. Allocating impacts with CFF led to lower impacts than economic allocation due different system boundaries, but the differences between the results for CFF allocation factors 1 and 0.5 were relatively small. In CFF, the feedstock is handled as waste and no burdens are allocated to feedstock from production, but allocation of the recycling process is done between the system producing the waste and the system using the waste material. Therefore, the results with this method are somewhere between handling materials as a product and a residue.

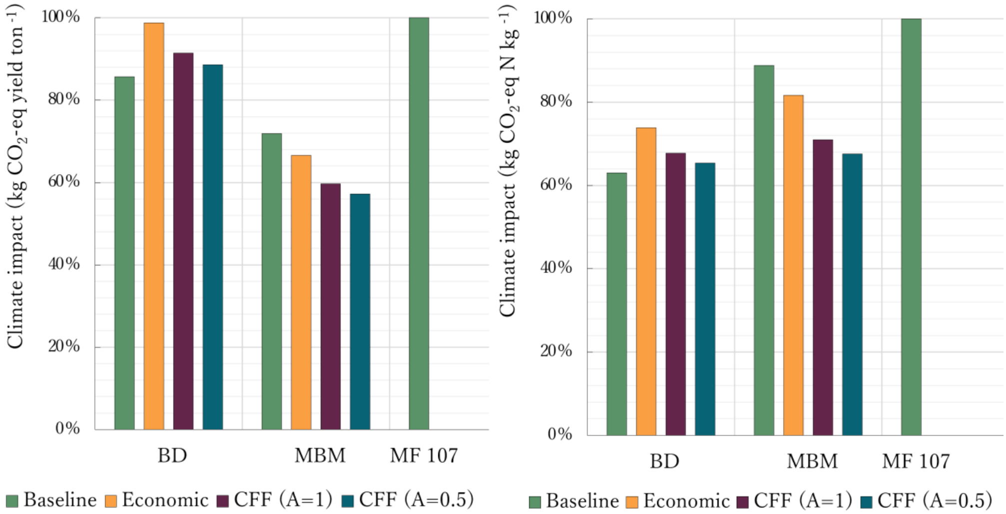

Fig. 9. *Relative climate impact of fertilisers with broader system boundaries and different allocation methods, with raw materials handled as residues in the baseline, as products in economic allocation and as wastes in CFF. A = allocation factor between the supplier and user of recycled materials (1 = all to user) in CFF, BD = biogas digestate, CFF = Circular Footprint Formula, MBM = meat and bone meal, MF 107 = mineral fertiliser (107 kg N ha^{-1}).*

4.3 Avoiding allocation through system expansion (Paper III)

Using system expansion instead of allocation resulted in the dairy cattle system having the highest impacts across all the impact categories, with over six times the land use impacts and over two times the global warming impacts of other systems (Fig. 10). Including the processing of animal by-products into different end products, such as hide into leather, barely increased the impacts of the animal systems, whereas adding the alternative products to the compared systems significantly increased their impacts. The milk alternative soy milk accounted for most of the impacts in pork, cultured meat and tofu systems.

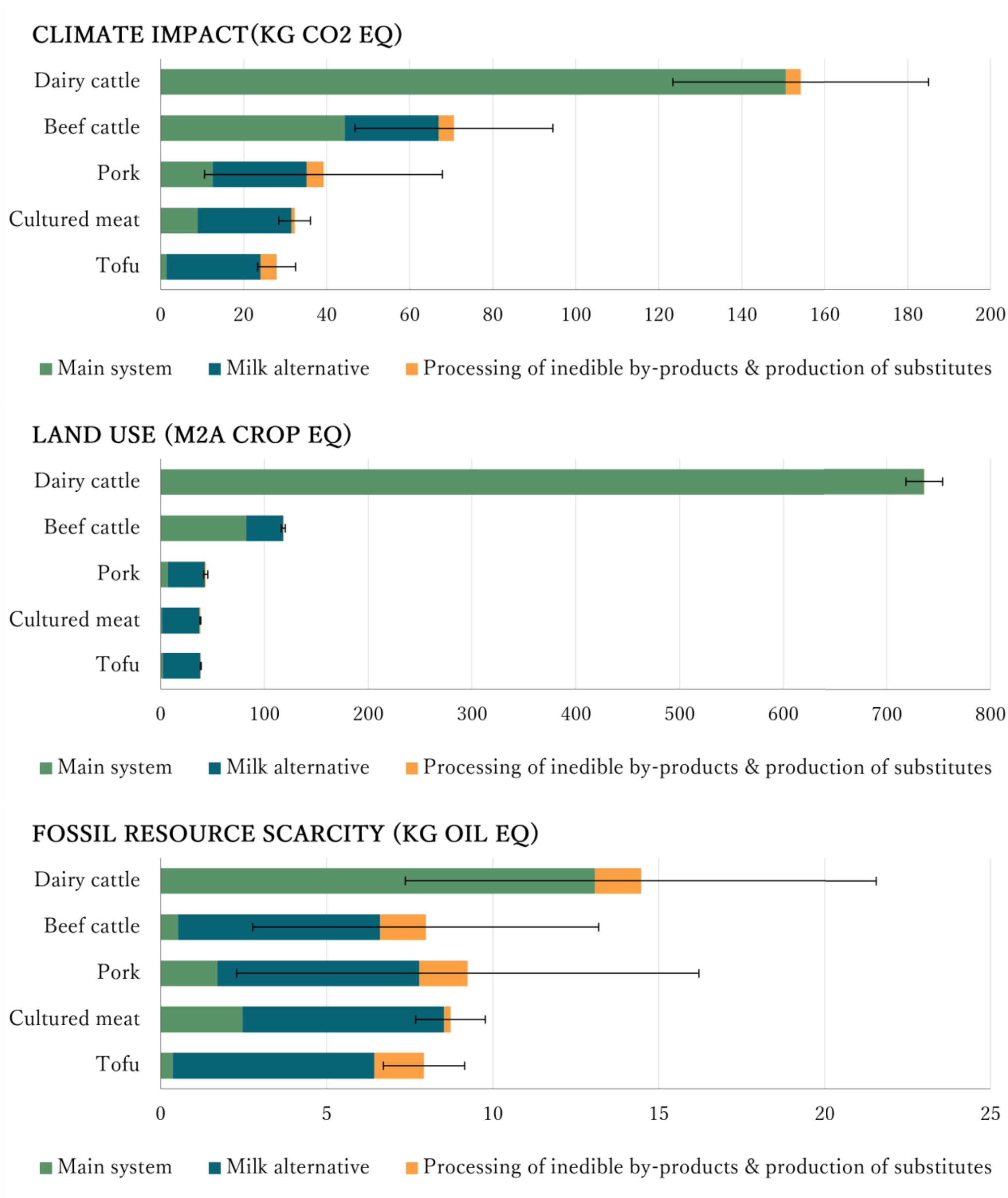

Fig. 10. *Climate impact (kg CO_2 eq.), land use (m^2a crop eq.) and fossil resource scarcity (kg oil eq.) of expanded dairy cattle, beef cattle, pork, cultured meat and tofu systems (expanded FU presented in Table 3). The main system includes all the unallocated impacts arising from the production of the main product, milk alternative the production of soymilk, and processing of inedible by-products & production substitutes all the impacts arising from*

further processing of the by-products or the impacts of producing equivalent alternative products. The error bars represent the standard deviation obtained with Monte Carlo analysis.

When the relative allocated product level results were compared with the system-level results, the climate impacts and land use of beef from dairy cattle and beef cattle were opposite (Fig. 11). Especially for beef meat originating from dairy cattle, the relative product level results are significantly lower than system level results. The fossil resource scarcity of cultured meat also differs considerably from system-level results, since the by-product lactic acid is formed in large amounts, but its economic value is low, leading to the majority of impacts being allocated to cultured meat.

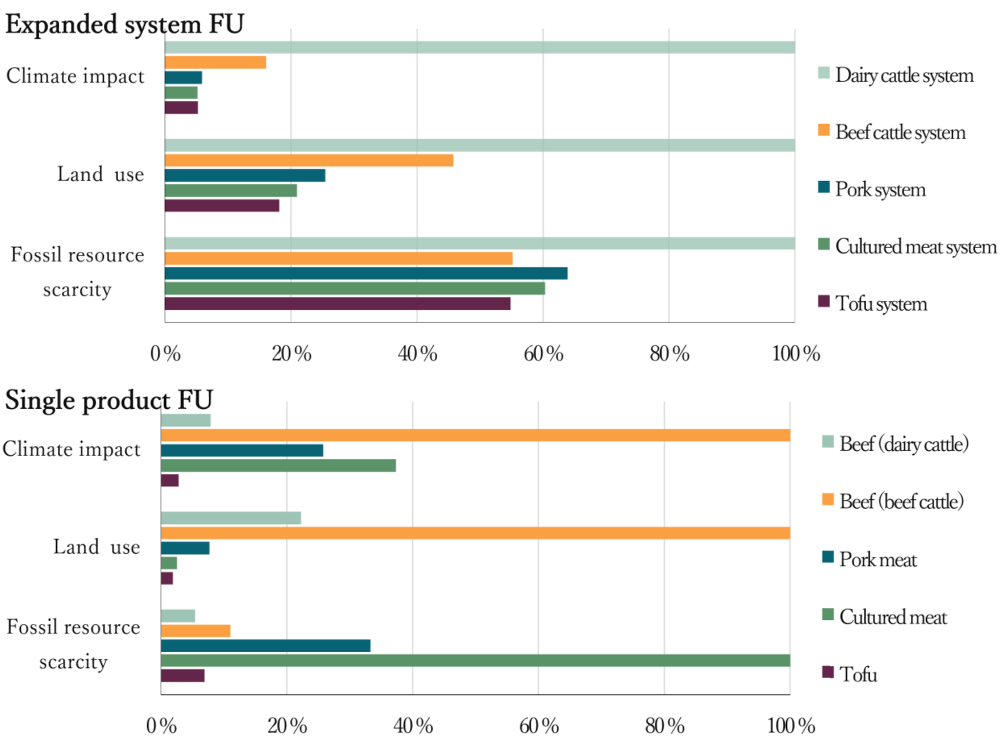

Fig. 11. *Relative climate impact, land use and fossil resource scarcity for the extended systems (Fig. 10, FU in Table 3) and for the FU of 1 kg of tofu, cultured meat, pork meat, and beef meat from dairy cattle and beef cattle, when using economic allocation.*

5. Discussion

5.1 Allocation in agricultural systems

With the assessment of measures of the circular economy, the need for proper product identification and handling in LCAs increases. Since all allocation methods have weaknesses and several methods are usually applicable to different systems (papers I–III), the choice of method must be critically evaluated when conducting an LCA study.

Agricultural systems can generate a variety of edible and non-edible materials, which can be used in a various way. Because they circulate in the system (e.g., manure or crop residues) or leave the farm without bringing in revenue (e.g., inedible animal body parts), some of these materials are not generally classified as products and handled as such (paper I). Inedible materials are also not identified as recyclable and no methods to handle recycling are applied (paper I).

The popularity of economic allocation and farm-gate system boundaries might complicate the recognition and classification of a variety of materials as by-products, since often they do not have economic value (papers I–III). To avoid this, the inclusion of downstream processes in the inventory and then allocation or system expansion at the point of substitution could be used.

Because the allocation method inevitably has an impact on the results, the method needs to be chosen in accordance with the study objectives. Especially in studies comparing different products from diverse production systems, the allocation method should enable a fair comparison. As shown in paper III, this is crucial in studies where the comparison includes main products from some systems and by-products from others (e.g., meat from beef cattle and meat from dairy cattle). Consequentially, the choice of allocation method is not as important in studies where the results are not used for comparison with other products but, for example, to identify the hotspots in the production chain.

5.2 Strengths and weaknesses of different methods

Primarily, allocation should be avoided whenever possible through sub-division processes or by system expansion (ISO 14040:2006). System expansion (as expanding the system) is not a widely used method, since it is not possible to obtain results for single products, and for comparison purposes the results are only usable in the context of the study in question (paper III).

The advantage of system expansion is that it takes into account variations in the mass, usage and quality of by-products, is unaffected by economic factors, and does not break the linkages of physical systems. System expansion considers a broader perspective of the production systems and treats compared systems more equally. Hence, it could also bring a new perspective to modelling the open-loop recycling of materials in systems of the circular economy. However, the method might be challenging to apply through the life cycle and considerable uncertainty is related to the implementation (paper III). Although system expansion includes an alternative way of producing materials that are absent from compared systems, it does not capture the consequential impacts of choosing one system over another, and thus cannot be used as a consequential assessment.

The chosen allocation method should consider the possible differences in the properties of the products, such as quality or usage. Biophysical allocation is based on physical causality and therefore reflects how the inputs of a process are utilized by each output. Therefore, in assessments of circular systems, biophysical allocation could be an appropriate method to address the impacts through various life cycles. Independence from economic values would also favour the use of the biophysical method for products of the circular economy.

In the ISO standard, biophysical allocation is preferred over other allocation methods, such as economic or mass allocation. However, it should be noted that earlier, Lindfors et al., (1995) presented allocation based on economic/social causality as a separate step above physical parameters (such as mass or energy), whereas the ISO standard includes both of these options in the last step.

The biophysical allocation method is justified and independent of variation in external factors, such as price, since it is solely based on the underlying physical relationship between materials. On the other hand, the method does not take a stand on why the production process exists in the first place (i.e., what the main product is) or address the varying quality of the different by-products. For example, in the case of animals to be slaughtered, this leads to a notably lower share of impacts being allocated to meat and higher impacts to slaughtering by-products than other allocation methods (paper I; Al-Zohairi et al., 2022; Chen et al., 2017). In addition, the causal physical relationships of biological processes are complex interactions and thus challenging to convert into a simple allocation formula. Ijassi et al. (2021) suggested using sensitivity analyses to find the causal relationship between by-products, but the approach is not feasible for biological processes, since the individual outputs of plant or animal production cannot be changed without also affecting the other outputs (Mackenzie et al., 2017). Another challenge in biophysical allocation is that the causal effect of some inputs (for example, heating and ventilation of buildings) is difficult to determine and relate to metabolic energy flows, even though they may affect the biological processes of animals (Mackenzie et al., 2017).

Although biophysical allocation is based on physical causality, at least in the case of dairy production, it uses economic value as a criterion to determine which output materials are considered as products with impacts allocated to them (Mackenzie et al., 2017). In this sense, the currently used biophysical allocation manner is not completely based on physical causality and detached from the socioeconomic context, since materials classified as waste might vary depending on the economic and cultural circumstances (Chen et al., 2017). Another weakness of the biophysical allocation method is that no such method has been developed for all products, and in studies comparing different products it might therefore be questionable to use one allocation method for some products and another method for others.

Economic allocation might be preferred, since it is easy to apply and illustrate the properties of complex systems, for example also considering differences in output material qualities (Ardente & Cellura, 2012). The strength of the method is that it is relatively easy to apply for all products in multiple stages of the life cycle and is also easily understood outside the LCA community. The considerable weakness of economic allocation is the dependency on the temporal and spatial context. Paper II demonstrated that economic allocation might lead to materials that are utilized being incorrectly classified as wastes when they do not have economic value. Furthermore, as a consequence of economic allocation, the environmental impacts vary depending on the demand for the product, even if the physical emissions from the production chain remain the same. Therefore, economic allocation might not be feasible to assess products of the circular economy. However, despite the defects of biophysical and economic allocation, these methods are usually preferred by LCA practitioners (Paper I; Wilfart et al., 2021).

Paper I discussed how protein and energy allocations are related to the fundamental purpose of dairy and beef production as the provider of nutrition, which makes them a relevant basis for allocation between milk and beef but not relevant for inedible by-products, which have other functions. Thus, allocation based solely on nutritional functions is not appropriate when also considering perspectives of the circular bioeconomy. In addition, including nutrition in LCA through one nutrient or energy content is in general too simplified an approach from the nutritional perspective (Mclaren et al., 2021; Saarinen et al., 2017).

Mass allocation is solely based on the masses of outputs and environmental impacts are allocated accordingly. The method is easy to use, since information concerning the volumes of different outputs is anyway needed to perform the assessment. Svanes et al. (2011) stated that mass allocation makes visible the differences caused by different system boundaries or physical differences in the value chain and does not fluctuate as economic allocation does. They suggest that mass allocation should be preferred for external communication to the market. The weakness of the method is that it is not able to reflect any causality, usage or quality of the materials. Hence, mass allocation should only be used when there is no difference in these above-mentioned characteristics, because otherwise it might favour main products at the expense of materials of less importance. Although mass allocation is a rarely used method, in the context of the circular bioeconomy it could be justified in situations where there is lack of data on the characteristics of different by-products.

5.3 Impact of allocation method on the results

The results of papers I–III underline the importance of the choice of allocation method. Papers I & II demonstrated that the results of LCAs can be highly dependent on the allocation method chosen; the share of impacts allocated for meat varied from 27% to 100%, depending on the method, and decreased the emissions by up to 24% compared to the baseline for recycled fertilizers. In turn, for milk, the variation was notably lower, from 82% to 100% in the reviewed milk LCA studies in paper I.

In paper I, it was found that system expansion is only implemented as substitution in milk production LCAs by subtracting the impacts arising from beef production from a beef herd from the milk production system. On average, this method leads to 61% of impacts being left for milk. As shown in paper III, applying system expansion as expanded system boundaries leads to the opposite results, demonstrating that the production of beef from a beef herd and milk from plant-based sources has lower environmental impacts than the dairy system. Hence, comparison assessments performed with system expansion differ from allocated single product results, especially if the system produces large volumes of by-products or by-products with a relatively high economic value (paper III).

Using the substitution method creates systems that do not represent a situation that could exist in reality: only obtaining milk from dairy production and meat from beef herds. Applying substitution to the attributional model also easily results in negative environmental impacts (e.g., Colley et al., 2020). Furthermore, allocation artificially separates physical processes, creating a situation where products are viewed separately, even though decisions made concerning one product inevitably also affect the other product. Hence, expanded system boundaries more realistically represent the physical reality.

However, following this same logic and the theory presented by Tillman et al. (1994), substitution could be applied to the dairy production system when comparing milk with plant-based alternatives by subtracting beef production (from beef herds) from the systems and comparing meat from dairy or beef cattle by subtracting the production of milk alternatives from the dairy production system.

5.4 Dependence on other methodological choices

LCA practitioners must make several choices that define how the method is applied (Fig. 12). Firstly, the study goal affects whether the modelling approach is attributional or consequential. In consequential modelling, multifunctionality is solved through substitution, while attributional modelling allows multiple methods to be used. Secondly, the function under study determines what kind of functional units and system boundaries are used. This affects the perceptions of what is the fundamental purpose of the system, which processes are linked to the life cycle and what type of multifunctionality is considered (e.g., material or immaterial functions). In addition, most databases used in LCA modelling contain pre-allocated data, such as the ecoinvent database, which is based on economic allocation. Hence, some of the upstream allocation choices are already made and the ones made by the LCA practitioner should take this into account. The effect of these choices is further discussed in following sections.

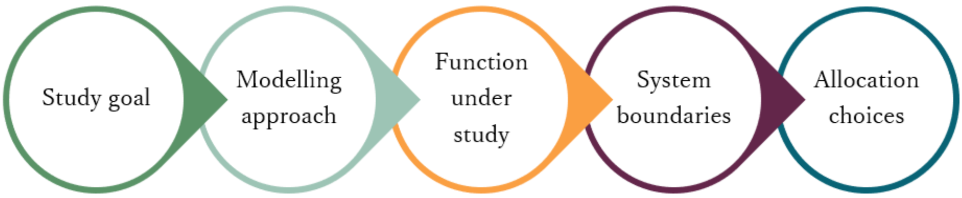

Fig. 12. Choices made in LCA affecting how the method is applied.

5.4.1 Study goal

All the choices, such as allocation, made in LCA should be in accordance with the study goal (ISO, 2006; Ijassi et al., 2021; Schrijvers et al., 2021). For example, if the study goal is to compare different products, the allocation method should be applicable to all the compared products through their life cycle. Biophysical allocation methods have only been developed for some products, which might lead to using some methods that are lower in allocation steps (ISO, 2006) in studies comparing several products (Paper I). As the by-products formed during the life cycle of different products usually vary in their quality, economic allocation might be feasible in comparison studies, since it is generally easy to apply and considers the differences in the quality of materials.

Studies only focusing on one system (e.g., finding hotspots) without comparison with other products do not necessarily even need to obtain the results for single products, and can thus avoid allocation entirely. This was the case, for example, in some of the milk LCA studies reviewed in paper I.

System expansion is a suitable method for comparing broader entities than individual products, as shown in paper III and in a similar farm level-comparison study by Röös et al. (2016). This approach might produce new information especially in comparisons where one product originates from a multifunctional system and another from a monofunctional system. The method may also be used to identify opportunities for the circular economy to reduce environmental impacts and thus also help in political decision making. In addition, system expansion could also be used in sensitivity analyses as an alternative for allocation to examine the impact of allocation.

5.4.2 Modelling approach

Paper I demonstrated that system expansion is applied in attributional milk and beef LCA studies only as substitution. Even though the ISO standard does not differentiate consequential and attributional LCA, handling multifunctionality through substitution is the core concept of consequential modelling and should only be used in consequential assessments (Brander & Wylie, 2011; Pelletier et al., 2015; Weidema, 2003). Therefore, the modelling approach corresponding to the study goal also influences the choice of allocation method.

Consequential models examine cause–effect chains, in which case it is necessary to include all the products that are linked through the production chain. In this way, it is possible to examine wider effects resulting from changes in the demand for the product under study (Weidema, 2003). In attributional models, only system expansion or allocation should be applied, since attributional assessments should only include emissions that are actually caused by the system (Brander & Wylie, 2011), not the ones that are possibly avoided. Due to these fundamental differences between consequential and attributional modelling approaches, recycling should not be handled with a substitution approach in attributional models. However, many end-of-life equations include the calculation of credits from avoided primary production (Allacker et al., 2014).

5.4.3 System boundaries

Especially for recycled materials, the choice of system boundaries is significant, since they define which processes are part of the production system of recycled material (Schrijvers et al., 2016; paper II) and impacts can only be allocated from as far as systems are included. For example, handling recycling through the APOS approach requires the recycling process to be included in the system under study. However, it should be noted that LCAs of food products typically do not cover the life cycle stages after the consumer, and waste treatment and the possible recycling of nutrients are thus excluded.

Furthermore, as paper I demonstrated, in the assessments of foods, cradle-to-farm-gate system boundaries are common, even though the products may separate in a later stage (e.g., meat from the animal). By doing so, the inedible by-products are neglected, since it is the whole animal, not just the meat, that leaves the farm gate. To avoid inconsistency, the allocation should only be applied to products that are physically separated from each other.

In turn, system expansion requires the system boundaries to also include alternative production of by-products. In principle, allocation is not performed when using system expansion, but complex systems cannot be expanded endlessly, and allocation for background processes might be needed (paper III).

5.4.4 Functions of systems

Because the impacts are only allocated between products, it is important to define which outputs of the process are considered as products and which are classified as waste (or residue). The allocation method chosen in a study inevitably affects which outputs of a process are classified as products and which are handled as waste or residue. As the impacts arising from waste management should also be allocated to products, the classification of outputs might significantly affect the results. Economic value is often used as a criterion when defining whether a material is a waste or product (European Commission, 2010). As paper II demonstrated, the economic value might notably fluctuate, and especially if there are new utilizations for the material, the changing classification of the material adds considerable uncertainty to the results. Since the classification of output products is already an allocation choice, the classification could also follow the allocation steps given in the ISO standard (ISO 14040:2006, 2006; Leip et al., 2019). When classifying the outputs as products or wastes, a causal criterion could be used instead of an economic one, by defining whether the presence or absence of this product directly affects some product on the market (e.g., if no feedstock to anaerobic digestion, no biogas).

In addition to allocation, the choice of the functional unit as the "quantified performance of a product system" also reflects the considered function of the system (ISO, 2013). The allocation method must be chosen in relation to the functional unit, as some functional units (e.g., land area) may even eliminate the need for allocation (Schau & Fet, 2008). In this sense, a functional unit (e.g., a certain amount of protein) could also be seen as the first choice of allocation, as it defines the system functions to which the impacts are assigned. In addition, allocation and the functional unit are also related and both should be selected in accordance with the goal and scope of the study.

As discussed in paper III, when comparing different systems by using system expansion, the most critical step in implementing the method is defining the functions and thus the alternative products. Therefore, when defining the alternative products, it should be considered that the product may have several functions, and a mixture of products might be needed to replace all the functions (a market of functions substituted with different markets of functions).

The functionality of food as a provider of nutrition is also included in the food LCAs to an increasing extent as methodological approaches for nutritional LCA have been developed (McLaren et al. 2021). So far, nutrition-based functional units have mostly been used in ALCA studies comparing individual products, but some system-level assessments have also been carried out (e.g., Röös et al., 2016). The challenge with nutrient content-based comparisons is that unification of the functional unit at the level of individual nutrients easily leads into endless system expansion, where the functional unit needs to be supplemented with other products to make the nutrient content of each nutrient equal.

To overcome this issue, nutrient indexes including several nutrients in the form of a single score are used as the functional unit (Saarinen et al., 2017; McLaren et al., 2021). However, the weakness of this method is that products with the same nutrient index score may have completely different nutrient contents, and thus do not provide the same function in terms of individual nutrients. This is also why the systems compared in paper III were unified based on the masses of the products instead of the nutritional contents.

In addition to the provision of food and income for farmers, agricultural systems are also functioning ecosystems, providing other important benefits through supporting, regulating and cultural services (Millennium Ecosystem Assessment, 2003). Since LCA generally focuses on physical flows, immaterial benefits such as ecosystem services are not usually covered in LCA studies, but the few studies conducted on the topic indicate that the results obtained with the inclusion of ecosystem services as outputs differ greatly from the results of a standard LCA (Boone et al., 2019; Kiefer et al., 2015). Immaterial benefits can also be included as functions in system-level studies using system expansion, as demonstrated in a study by Röös et al. (2016).

5.5 Harmonisation of the method

Efforts have been made to harmonise the LCA method, especially to contain reliable and comparable product-specific information (European Commission, 2013). Unavoidably, one subject of harmonisation is allocation, for which purpose system- and product-specific recommendations are made (European Commission, 2018; Ijassi et al., 2021; Schrijvers et al., 2016, 2021). Furthermore, the SETAC/ACLCA Interest Group on Circularity & LCA is working to further clarify the methodical handling of circularity in LCA (Saidani et al., 2022). Thus, harmonized guidelines, such as PEFCRs, can help in choosing the allocation method and increase the comparability of studies. Since the allocation method should be chosen in accordance with the study goal, harmonization of the method is only feasible within assessments having the same study goal, and no strict recommendations covering all LCAs can be given.

5.6 Future research needs

Novel foods and their production technologies, such as cellular agriculture, are being developed to reduce the environmental impacts of food systems. Some of these technologies can both utilize and produce by-products (Smetana et al. 2015), which emphasizes the need for proper handling of by-products in LCAs. As paper III demonstrated, the relative system-level results might differ greatly from the allocated single product results, and system-level research should thus be performed to evaluate the assessment method as well as compare different systems.

It should also be considered that by-products are limited resources, and their availability is dependent on the production volumes of the main products. Therefore, changes in the consumption of one product might also have wider impacts, as demonstrated in paper III. Moreover, many by-products, such as the animal by-products presented in paper III, are already efficiently utilized, which should be taken into account when assessing alternative uses for these products (Sandström et al., 2022). Following the principles of the circular economy, waste can be turned into valuable resources, which emphasizes the importance of material availability.

Considering a wider perspective than impacts related to a single product is especially important when LCA results are used to guide decision making (Frehner et al., 2020). The linkage between products should be more often covered when comparing potential reductions in environmental impacts achieved by choosing one product over another. For example, studies aiming to optimize diets for minimum environmental impacts and adequate nutrition usually handle dairy products and beef separately (e.g., Chaudhary & Krishna, 2019; Mazac et al., 2022). Optimization based on single product LCA results leads to diets that contain dairy products but no meat (e.g., Mazac et al., 2022), which entirely neglects the underlying systems and thus incorrectly implies greater reductions in environmental impacts. A shift from animal source foods to plant-based foods also requires alternative production of the products that are currently produced from the inedible parts of animals.

Originally, LCA was used for assessing industrial processes, and especially their energy use (European Environment Agency, 1997). Industrial processes are very controllable and predictable, with a certain amount of inputs leading to a rather standard amount of outputs. Even though agriculture undeniably has industrial features, it fundamentally consists of complex biological processes. The product-centric approach of LCA is not, at least currently, able to catch the role of nature in agricultural systems, but focuses on assessing the systems as industrial processes. Therefore, the land use and level of intensity of agriculture should be better represented. Since agricultural systems are functioning ecosystems, natural mechanisms such as biodiversity, ecosystem services and carbon sequestration should also be included in the assessments. These functions could additionally be handled as outputs of the systems, which would require a whole new perspective on allocation issues. Many of these functions fundamentally relate to land use and land use change (Newbold et al., 2016; Sun et al., 2022). Therefore, system-level assessments and comparisons based on land use should be advanced.

6. Conclusions

The main outcome of this thesis is that allocation is always an artificial separation of a product from the production system, and the environmental impacts of a single product from multifunctional systems are therefore rather theoretical. Hence, assessments of wider systems through consequential modelling, system expansion or using unallocated results should be more frequently considered. Methodologically, unallocated results could be handled, for example, as a sensitivity when interpreting the results and using them to support decision making, because product-specific information will continue to be needed and allocation cannot therefore be entirely avoided.

This dissertation demonstrated that even though the challenge of allocating impacts between milk and meat has been discussed for decades, the wider multifunctionality of agricultural systems is weakly considered in LCA research. As agriculture has a significant role in the circular bioeconomy, this issue should be tackled in LCAs by considering all the materials produced, whether they have economic value or not. To follow the biophysical reality of production systems, market information should be excluded from the assessments when possible. This practice would also follow the allocation hierarchy given in the ISO standard (ISO, 2006). Even though economic allocation is a generally accepted and widely used method, for this reason it may not be suitable for assessments of the circular economy. Furthermore, the cut-off approach used in some databases does not capture the underlying impacts and complexity of circular systems.

Multifunctionality should always be handled in a way that is suitable in the context of the study in question and in accordance with other modelling decisions. Since each method has its own weaknesses, harmonization of the method according to a specific study goal can help LCA practitioners in choosing the appropriate method.

Nevertheless, many issues in LCA related to the circulation of materials and services provided by agriculture remain open. Since products from multifunctional systems can only be separated on a conceptual level, system-level comparisons should be further investigated in order to promote the sustainability of the circular economy. Overall, agriculture's multifunctionality, in terms of both products and natural mechanisms, should be better understood and considered in future research.

References

Allacker, K., Mathieux, F., Manfredi, S., Pelletier, N., De Camillis, C., Ardente, F., & Pant, R. (2014). Allocation solutions for secondary material production and end of life recovery: Proposals for product policy initiatives. Resour Conserv Recycl 88, 1–12. https://doi.org/10.1016/j.resconrec.2014.03.016

Al-Zohairi, S., Knudsen, M.T. & Mogensen, L. Environmental impact of Danish pork—effect of allocation methods at slaughtering stage. (2022). Int J Life Cycle Assess. https://doi.org/10.1007/s11367-022-02089-y

Ardente, F., & Cellura, M. (2012). Economic Allocation in Life Cycle Assessment. J Ind Ecol 16, 387–398. https://doi.org/10.1111/j.1530-9290.2011.00434.x

Boone, L., Roldán-Ruiz, I., Van linden, V., Muylle, H., & Dewulf, J. (2019). Environmental sustainability of conventional and organic farming: Accounting for ecosystem services in life cycle assessment. Sci Total Environ 695, 133841. https://doi.org/10.1016/j.scitotenv.2019.133841

Brander, M., & Wylie, C. (2011). The use of substitution in attributional life cycle assessment. Greenh Gas Meas Manag 1, 161–166. https://doi.org/10.1080/20430779.2011.637670

Brockmann, D., Pradel, M., & Hélias, A. (2018). Agricultural use of organic residues in life cycle assessment: Current practices and proposal for the computation of field emissions and of the nitrogen mineral fertilizer equivalent. Resour Conserv Recycl 133, 50–62. https://doi.org/10.1016/j.resconrec.2018.01.034

Campbell, B. M., Beare, D. J., Bennett, E. M., Hall-Spencer, J. M., Ingram, J. S. I., Jaramillo, F., Ortiz, R., Ramankutty, N., Sayer, J. A., & Shindell, D. (2017). Agriculture production as a major driver of the Earth system exceeding planetary boundaries. Ecol Soc 22. https://doi.org/10.5751/ES-09595-220408

Chaudhary, A. & Krishna, V. (2019). Country-Specific Sustainable Diets Using Optimization Algorithm. Environ Sci Technol 53, 7694-7703. https://doi.org/10.1021/acs.est.8b06923

CE Delft. (2021). TEA of cultivated meat. Future projections of different scenarios.

Cederberg, C., & Stadig, M. (2003). System Expansion and Allocation in Life Cycle Assessment of Milk and Beef Production Int J Life Cycle Assess 8, 350–356. https://doi.org/10.1007/BF02978508

Chen, X., Wilfart, A., Puillet, L., & Aubin, J. (2017). A new method of biophysical allocation in LCA of livestock co-products: modeling metabolic energy requirements of body-tissue growth. Int J Life Cycle Assess 22, 883–895. https://doi.org/10.1007/s11367-016-1201-y

Colley, TA., Birkved, M., Olsen, S. I. & Hauschild, M. Z. (2020). Using a gate-to-gate LCA to apply circular economy principles to a food processing SME. J Clean Prod 251, 119566. https://doi.org/10.1016/j.jclepro.2019.119566

EDA. (2018). Product Environmental Footprint Category Rules (PEFCR) for Dairy Products.

European Commission. (2012). Innovating for sustainable growth - A Bioeconomy for Europe. https://op.europa.eu/en/publication-detail/-/publication/1f0d8515-8dc0-4435-ba53-9570e47dbd51

European Commission. (2013). Recommendation 2013/179/EU on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations. Official Journal of European Union. https://doi.org/doi:10.3000/19770677.L_2013.124.eng

European Commission. (2018). Product Environmental Footprint Category Rules Guidance. PEFCR Guidance Document, - Guidance for the Development of Product Environmental Footprint Category Rules (PEFCRs), Version 6.3.

European Commission. (2010). International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. In Constraints. https://doi.org/10.2788/38479

European Environment Agency. (1997). Life Cycle Assessment. A guide to approaches, experiences and information sources. Environmental Issues Series no. 6. https://www.eea.europa.eu/publications/GH-07-97-595-EN-C/Issue-report-No-6.pdf/view

FEDIAF. (2018). Product Environmental Footprint Category Rules. Prepared Pet Food for Cats and Dogs. www.quantis-intl.com

Flysjö, A., Cederberg, C., Henriksson, M., & Ledgard, S. (2011). How does co-product handling affect the carbon footprint of milk? Case study of milk production in New Zealand and Sweden. Int J Life Cycle Assess 16, 420–430. https://doi.org/10.1007/s11367-011-0283-9

Frehner, A., Muller, A., Schader, C., De Boer, I. & Van Zanten, H. (2020). Methodological choices drive differences in environmentally-friendly dietary solutions. Glob Food Sec 24, 100333. https://doi.org/10.1016/j.gfs.2019.100333

Gac, A., Lapasin, C., Laspière, P. T., Guardia, S., Ponchant, P., Chevillon, P., & Nassy, G. (2014). Co-products from meat processing: the allocation issue. Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2014), 438–442.

Gilardino, A., Quispe, I., Pacheco, M., & Bartl, K. (2020). Comparison of different methods for consideration of multifunctionality of Peruvian dairy cattle in Life Cycle Assessment. Livest Sci 240, 104151. https://doi.org/10.1016/J.LIVSCI.2020.104151

Hanserud, O. S., Cherubini, F., Øgaard, A. F., Müller, D. B., & Brattebø, H. (2018). Choice of mineral fertilizer substitution principle strongly influences LCA environmental benefits of nutrient cycling in the agri-food system. Sci Total Environ 615, 219–227. https://doi.org/10.1016/j.scitotenv.2017.09.215

Heijungs, R., Allacker, K., Benetto, E., Brandão, M., Guinée, J., Schaubroeck, S., Schaubroeck, T., & Zamagni, A. (2021). System Expansion and Substitution in LCA: A Lost Opportunity of ISO 14044 Amendment 2. Front Sustainability 40. https://doi.org/10.3389/FRSUS.2021.692055

Hetemäki, L., Hanewinkel, M., Muys, B., Ollikainen, M., Palahí, M., & Trasobares, A. (2017). Leading the way to a European circular bioeconomy strategy. From Science to Policy 5. https://doi.org/10.36333/fs05

Heusala, H., Sinkko, T., Sözer, N., Hytönen, E., Mogensen, L., & Knudsen, M. T. (2020). Carbon footprint and land use of oat and faba bean protein concentrates using a life cycle assessment approach. J Clean Prod 242, 118376. https://doi.org/10.1016/j.jclepro.2019.118376

Huijbregts, M. A. J., Steinmann, Z. J. N., Elshout, P. M. F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., & van Zelm, R. (2017). ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int J Life Cycle Assess 22, 138–147. https://doi.org/10.1007/s11367-016-1246-y

Humbird, D. (2020). Scale-Up Economics for Cultured Meat: Techno-Economic Analysis and Due Diligence. preprint. https://doi.org/10.1002/bit.27848

Huygens, D., & Saveyn, H. G. M. (2018). Agronomic efficiency of selected phosphorus fertilisers derived from secondary raw materials for European agriculture. A meta-analysis. Agron Sustain Dev 38, 1–14. https://doi.org/10.1007/S13593-018-0527-1

Ijassi, W., Ben Rejeb, H., & Zwolinski, P. (2021). Environmental impact evaluation of co-products: decision-aid tool for allocation in LCA. Int J Life Cycle Assess 1, 1–16. https://doi.org/10.1007/S11367-021-01984-0

Ineichen, S., Schenker, U., Nemecek, T. & Reidy, B. (2022). Allocation of environmental burdens in dairy systems: Expanding a biophysical approach for application to larger meat-to-milk ratios. Livest Sci 261, 104955. https://doi.org/10.1016/j.livsci.2022.104955

ISO 14040:2006. (2006). Environmental management. Life cycle assessment. Principles and framework (ISO 14040:2006).

JRC. (2016). Baseline Approaches for the Cross-Cutting Issues of the Cattle Related Product Environmental Footprint Pilots in the Context of the Pilot Phase.

Kiefer, L. R., Menzel, F., & Bahrs, E. (2015). Integration of ecosystem services into the carbon footprint of milk of South German dairy farms. J Environ Manage 152, 11–18. https://doi.org/10.1016/j.jenvman.2015.01.017

Leip, A., Ledgard, S., Uwizeye, A., Palhares, J. C. P., Aller, M. F., Amon, B., Binder, M., Cordovil, C. M. D. S., De Camillis, C., Dong, H., Vertès, F., & Wang, Y. (2019). The value of manure - Manure as co-product in life cycle assessment. J Environ Manage 241, 293–304. https://doi.org/10.1016/j.jenvman.2019.03.059

Lindfors, L.-G., Christiansen, K., Hoffmann, L., Virtanen, Y., Juntilla, V., Hanssen, O., Rønning, A., Ekvall, T., & Finnveden, G. (1995). Nordic Guidelines on Life-Cycle Assessment. Nord1995:20.

Mackenzie, S. G., Leinonen, I., & Kyriazakis, I. (2017). The need for co-product allocation in the life cycle assessment of agricultural systems—is "biophysical" allocation progress? Int J Life Cycle Assess 22, 128–137. https://doi.org/10.1007/s11367-016-1161-2

Mazac, R., Meinilä, J., Korkalo, L. et al. Incorporation of novel foods in European diets can reduce global warming potential, water use and land use by over

80%. (2022). Nat Food 3, 286–293. https://doi.org/10.1038/s43016-022-00489-9

Mclaren, S., Berardy, A., Henderson, A., Holden, N., Huppertz, T., Jolliet, O., Renouf, M., Rugani, B., Saarinen, M., Van Der Pols, J., Vázquez-Rowe, I., Vallejo, A., Bianchi, M., Chaudhary, A., Chen, C., Cooreman-Algoed, M., Dong, H., Grant, T., Green, A., … Van Zanten, H. (2021). Integration of environment and nutrition in life cycle assessment of food items: opportunities and challenges. FAO. https://www.fao.org/3/cb8054en/cb8054en.pdf

Millennium Ecosystem Assessment. (2003). Ecosystems and Human Well-being: A Framework for Assessment.

Möller, K., Oberson, A., Bünemann, E. K., Cooper, J., Friedel, J. K., Glæsner, N., Hörtenhuber, S., Løes, A. K., Mäder, P., Meyer, G., Müller, T., Symanczik, S., Weissengruber, L., Wollmann, I., & Magid, J. (2018). Improved Phosphorus Recycling in Organic Farming: Navigating Between Constraints. Adv Agron 147, 159–237. https://doi.org/10.1016/BS.AGRON.2017.10.004

Newbold, T., Hudson, L. N., Arnell, A. P., Contu, S., De Palma, A., Ferrier, S., Hill, S. L. L., Hoskins, A. J., Lysenko, I., Phillips, H. R. P., Burton, V. J., Chng, C. W. T., Emerson, S., Gao, D., Hale, G. P., Hutton, J., Jung, M., Sanchez-Ortiz, K., Simmons, B. I., … Purvis, A. (2016). Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. Science 353, 291–288. https://doi.org/10.1126/science.aaf2201

OSF. (2021). OSF. Official Statistics of Finland: Producer Prices of Agricultural Products. https://www.stat.fi/til/matutu/index_en.html

Pelletier, N., Ardente, F., Brandão, M., De Camillis, C., & Pennington, D. (2015). Rationales for and limitations of preferred solutions for multifunctionality problems in LCA: is increased consistency possible? Int J Life Cycle Assess 20, 74–86. https://doi.org/10.1007/s11367-014-0812-4

Poore, J., & Nemecek, T. (2018). Reducing food's environmental impacts through producers and consumers. Science 360, 987–992. https://doi.org/10.1126/science.aaq0216

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F. S., Lambin, E., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., De Wit, C. A., Hughes, T., Van Der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., … Foley, J. (2009). Planetary Boundaries: Exploring the Safe Operating Space for Humanity. Ecol Soc 14, 2. https://www.jstor.org/stable/26268316

Roos Lindgreen, E., Mondello, G., Salomone, R. et al. Exploring the effectiveness of grey literature indicators and life cycle assessment in assessing circular economy at the micro level: a comparative analysis. Int J Life Cycle Assess 26, 2171–2191 (2021). https://doi.org/10.1007/s11367-021-01972-4

Röös, E., Patel, M., & Spångberg, J. (2016). Producing oat drink or cow's milk on a Swedish farm — Environmental impacts considering the service of grazing, the opportunity cost of land and the demand for beef and protein. Agric Syst 142, 23–32. https://doi.org/10.1016/J.AGSY.2015.11.002

Saarinen, M., Fogelholm, M., Tahvonen, R., & Kurppa, S. (2017). Taking nutrition into account within the life cycle assessment of food products. J Clean Prod 149, 828–844. https://doi.org/10.1016/j.jclepro.2017.02.062

Saidani, M., Kreuder, A., Babilonia, G. et al. (2022) Clarify the nexus between life cycle assessment and circularity indicators: a SETAC/ACLCA interest group. Int J Life Cycle Assess 27, 916–925. https://doi.org/10.1007/s11367-022-02061-w

Sandström, V., Chrysafi, A., Lamminen, M. et al. (2022) Food system by-products upcycled in livestock and aquaculture feeds can increase global food supply. Nat Food. https://doi.org/10.1038/s43016-022-00589-6

Schau, E. M., & Fet, A. M. (2008). LCA studies of food products as background for environmental product declarations. Int J Life Cycle Assess 13, 255–264. https://doi.org/10.1065/lca2007.12.372

Schrijvers, D. L., Loubet, P., & Sonnemann, G. (2016). Developing a systematic framework for consistent allocation in LCA. Int J Life Cycle Assess 21, 976–993. https://doi.org/10.1007/s11367-016-1063-3

Schrijvers, D. L., Loubet, P., & Sonnemann, G. (2021). An axiomatic method for goal - dependent allocation in life cycle assessment. Int J Life Cycle Assess 1, 3. https://doi.org/10.1007/s11367-021-01932-y

Smetana, S., Mathys, A., Knoch, A. et al. (2015). Meat alternatives: life cycle assessment of most known meat substitutes. Int J Life Cycle Assess 20, 1254–1267. https://doi.org/10.1007/s11367-015-0931-6

Spångberg, J., Hansson, P.-A., Tidåker, P., & Jönsson, H. (2011). Environmental impact of meat meal fertilizer vs. chemical fertilizer. Resour Conserv Recycl 55, 1078–1086. https://doi.org/10.1016/j.resconrec.2011.06.002

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., De Vries, W., De Wit, C. A., Folke, C., Gerten, D., Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., & Sörlin, S. (2015). Planetary boundaries: Guiding human development on a changing planet. Science 34, 6223. https://doi.org/10.1126/science.1259855

Stegmann, P., Londo, M., & Junginger, M. (2020). The circular bioeconomy: Its elements and role in European bioeconomy clusters. Resour Conserv Recycl 6, 100029. https://doi.org/10.1016/J.RCRX.2019.100029

Sun, Z., Scherer, L., Tukker, A., Spawn-Lee, S. A., Bruckner, M., Gibbs, H. K., & Behrens, P. (2022). Dietary change in high-income nations alone can lead to substantial double climate dividend. Nat Food 2022, 1–9. https://doi.org/10.1038/s43016-021-00431-5

Svanes, E., Vold, M., & Hanssen, O. J. (2011). Effect of different allocation methods on LCA results of products from wild-caught fish and on the use of such results. Int J Life Cycle Assess 16, 512–521. https://doi.org/10.1007/S11367-011-0288-4/TABLES/5

Thoma, G., Jolliet, O., & Wang, Y. (2013). A biophysical approach to allocation of life cycle environmental burdens for fluid milk supply chain analysis. International Dairy, 31. https://doi.org/10.1016/j.jclepro.2011.11.046

Tillman, A. M., Ekvall, T., Baumann, H., & Rydberg, T. (1994). Choice of system boundaries in life cycle assessment. Journal of Cleaner Production, 2(1), 21–29. https://doi.org/10.1016/0959-6526(94)90021-3

Weidema, B. (2003). Market information in life cycle assessment. Environmental Project No. 863.

Weidema, B. (2014). Has ISO 14040/44 Failed Its Role as a Standard for Life Cycle Assessment? Journal of Industrial Ecology, 18(3), 324–326. https://doi.org/10.1111/jiec.12139

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The Ecoinvent database version 3. In The International Journal of Life Cycle Assessment. http://link.springer.com/10.1007/s11367-016-1087-8

Wilfart, A., Gac, A., Salaün, Y., Aubin, J., & Espagnol, S. (2021). Allocation in the LCA of meat products: is agreement possible? Cleaner Environmental Systems, 2, 100028. https://doi.org/10.1016/J.CESYS.2021.100028