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MASTER'S THESIS

A prospective and context-based modelling approach: Finding suitable Circular Economy scenarios for bio-based side and waste streams in Hämeenkyrö, Finland

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Content

Abbreviations

AD	Anaerobic Digestion
AESA	Absolute Environmental Sustainability Assessment
CBE	Circular Bioeconomy
CE	Circular Economy
CF	Characterisation Factor
EF	Environmental Footprint Method
EM	Electronic Marketplace
EoL	End-of-Life
FU	Functional Unit
GHG	Greenhouse Gas
IAM	Integrated Assessment Model
LCA	Life-Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MFA	Material Flow Assessment
NPi	National Policies Implemented
PB	Planetary Boundary
pLCA	Prospective LCA
SAF	Sustainability Assessment Framework
SSP	Shared Socio-economic Pathway

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1 Introduction

In recent decades, human civilization has significantly raised its consumption of finite resources and is increasingly stressing the limits of the planetary boundaries (PB). A key reason for this is a lack of innovation regarding the utilization of alternative regenerative resources and the underdevelopment of regional biowaste markets. Additionally, conventional treatment of biowaste further contributes to environmental degradation through harmful emissions.

By implementing circular economy (CE) scenarios, regenerative resources can be used as a resource in new value chain.

Therefore, this thesis seeks to determine the environmental potential of implementing an electronic marketplace (EM), thus increasing the profitability of CE value chains.

At large scales, bioenergy systems are a commonly available CE pathway. These are known to generally improve the environmental footprint of conventional biowaste treatment. However, the individual environmental impacts of bioenergy are dependent on the regional context of the study object.

Regionalized biowaste market data are commonly not available but are made accessible for the municipality of Hämeenkyrö in Finland. Therefore, this thesis conducts a case study to assess the sustainability of four bioenergy pathways in Hämeenkyrö.

For that purpose, the life cycle assessment (LCA) methodology is adapted to account for the specific regional context. This dedicated methodology is used to answer the following research question:

“To what extent does employing new bioenergy pathways reduce the PB depletion of biowaste treatment in the Hämeenkyrö municipality over the next decades?”

This case study is part of the scientific effort to find context-specific sustainability assessment methods of CE scenarios. Therefore, this thesis also evaluates the employed novel methodological approaches and reflects on their suitability in the case study’s context. Recommendations for future methodological adaptations are made. This case study is part of the research project TREASoURcE by the European Union, which aims at supporting the development of CE pathways.

In the following chapter, the current state of research on the above-mentioned conceptual frameworks and topics is introduced. In the third chapter, the dedicated context-based sustainability assessment framework (SAF) is established. The results of this case study are presented in the fourth chapter. Then, the results are evaluated and discussed with a focus on the evaluation tools of the LCA methodology. Finally, policy recommendations and conclusions are drawn.

2 State of Research

This chapter introduces the current state of research on key scientific concepts of this case study. First, key scientific concepts are introduced. Secondly, the current state of biowaste treatment pathways in Finland is outlined, and two potentially suitable bioenergy technologies are described. Thirdly, an introduction into LCA is provided. Finally, five hypotheses are formulated for this case study.

In this thesis, the term "biowaste" is used synonymously with the phrase "bio-based side and waste streams" to describe biological residues e.g. from agricultural production, or from households.

2.1 Conceptual Frameworks

This chapter introduces the following four concepts, which are essential for this thesis' case study: PB, CE, and EM.

2.1.1 Planetary Boundaries Framework

Since the industrial era, human activities have started to threaten the functionality of the Earth system. Therefore, Rockström *et al.* (2009) identified nine absolute planetary safe operating spaces of human activities. Each safe operating space represents a critical threshold that, if breached, causes unacceptable change to one of Earth's critical subsystems. So-called control variables measure the current state of PB's depletion.

Exceeding a PB is argued to potentially trigger tipping points that will disrupt human production and consumption patterns. Limiting global emissions to remain inside these threshold is argued to promote the achievement of the social development goals (Sala *et al.* 2020).

Currently, seven out of these nine PBs are exceeded, namely Climate Change, Change in Biosphere Integrity, Land System Change, Freshwater Change, Modification of Biogeochemical Flows, Introduction of Novel Entities, and Ocean Acidification. Only two PBs are currently within their respective safe operating space: Increase in Atmospheric Aerosol Loading and Stratospheric Ozone (Sakschewski *et al.* 2025).

2.1.2 Circular Economy Scenarios

The concept of CE refers to a broad economic scenario. There are many definitions of CE. A definition by the European Commission (2019) encompasses the retention of material values, the minimisation of waste and resource use as well as the reuse of resources, among other characteristics.

The goals of CE include the strengthening of resource efficiency of processes and to reduce the use of fossil carbon (Carus and Dammer 2018). For that purpose, various CE scenarios and levels of circularity have been defined. The following figure illustrates the “R9-Scenarios” of the CE, as described by the European Commission (2020).

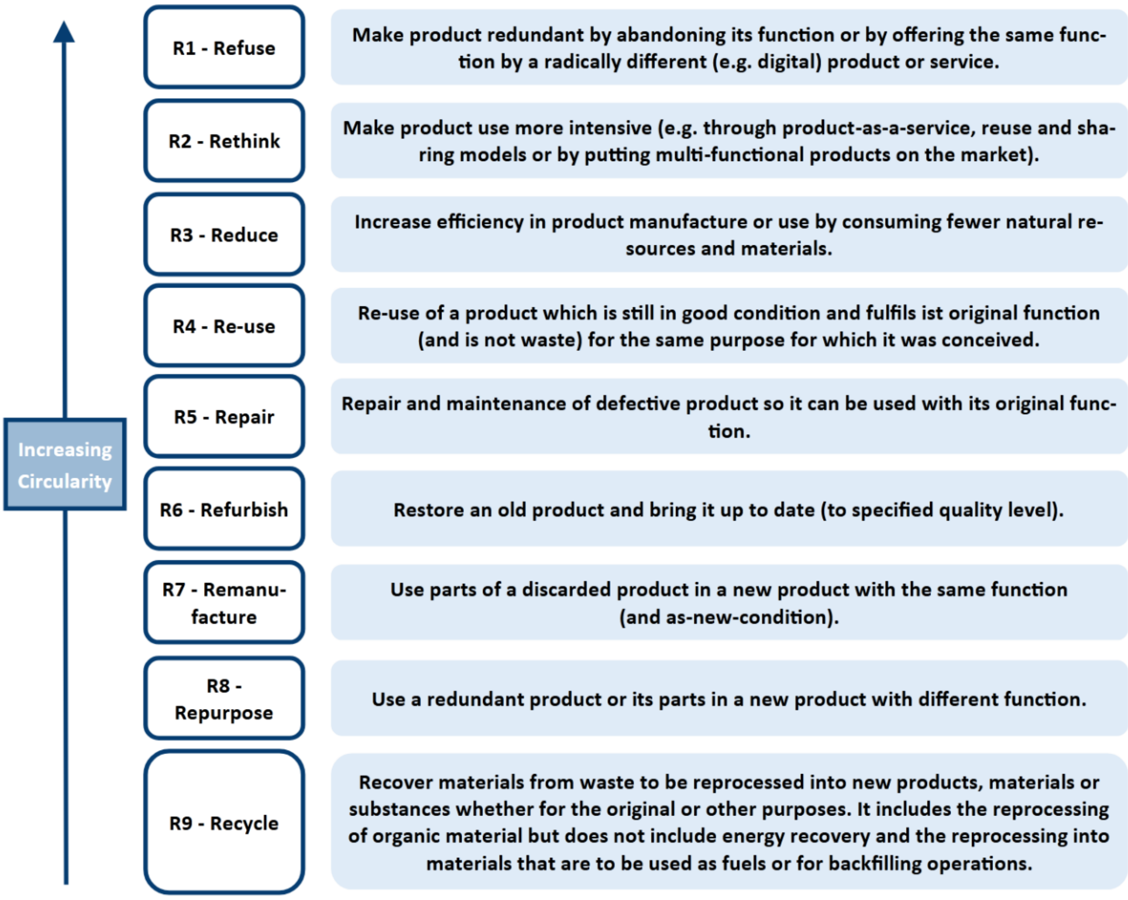

Figure 1: The R-Scenarios of Circular Economy.
Source: Based on European Commission (2020), Design adapted from Potting *et al.* (2017).

The exact definitions of the R-scenarios vary slightly between literature source (Potting *et al.* 2017). However, all definitions share a hierarchy of increasing circularity ranging, from recycling to refusal.

This definition by the European Commission (2020) explicitly excludes energetic recovery of organic materials from the scenario R9 – Recycle. They argue that the circularity of bioenergy is too limited compared to the other a core CE scenario (European Commission 2020). Therefore, this special case of biological materials in the CE framework is examined more closely.

Circular Bioeconomy Scenarios

Biological materials are especially relevant to the CEs goal of reducing the usage of fossil carbon, as they can substitute fossil carbon as a material. Scholars reference this concept as the bioeconomy (Carus and Dammer 2018).

Since the CE and bioeconomy share common goals like lower carbon footprints and resource efficiency, they can be seen as complementary approaches (Carus and Dammer 2018: p.4). The term circular bioeconomy (CBE) is used for conjunctions of these concepts (Salvador *et al.* 2021).

In addition to the characteristics of the CE, some specific characteristics of the CBE are the goal of maintaining material value, and the optimization of value over time is a key characteristic of CBE (Stegmann *et al.* 2020).

The CBE is commonly characterized by a high abundance of low material value. The concept of material value and resource quality are commonly interlinked with their range of suitable valorisations pathways. It should be emphasized that no obvious connection between material value and economic value exists. Instead, the economic value is context specific (Stegmann *et al.* 2020).

In reference to the CE-scenarios, a specified hierarchy for biomass valorisation can be identified, which is presented in Figure 2.

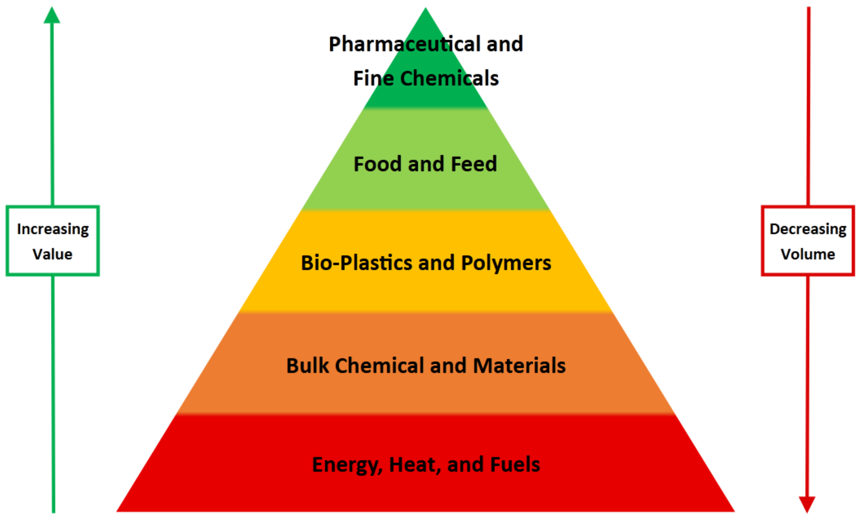

Figure 2: Visualisation of the bio-based value pyramid.
Source: Based on Stegmann *et al.* (2020), design adapted.

The hierarchy illustrates the decreasing abundance of suitable biomass for higher valorisation pathways. Therefore, biomasses should generally be utilised in the highest available pathway tier.

Valorisation can be further optimised by adding sequential usage of the biomass over time. Usually, a decrease in material value occurs. This is also referred to as cascading (Stegmann *et al.* 2020). The following Figure 3 visualises this concept.

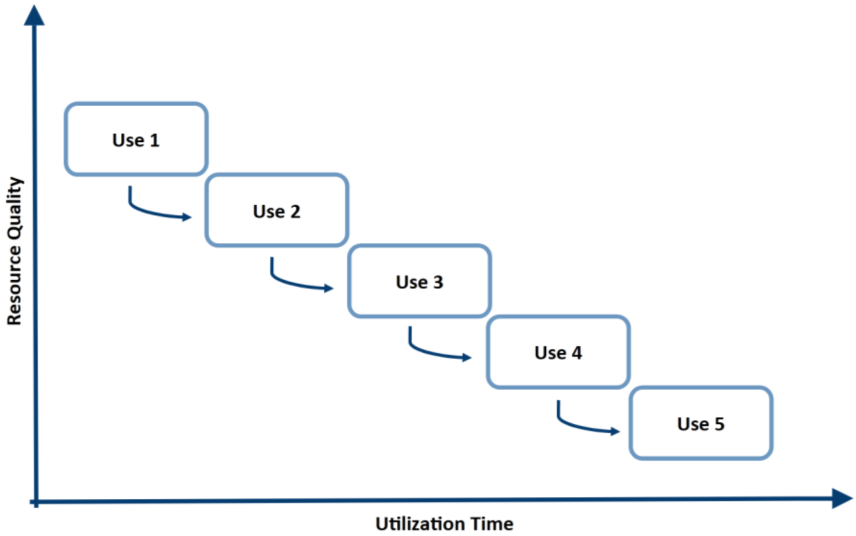

Figure 3: Visualisation of the cascading concept.
Source: Based on Stegmann *et al.* (2020), design adapted.

According to the R9-scenarios, raw manure field application can be assigned to the scenario R8-Repurpose. However, the concept of CE is ambiguous regarding the role of waste-based bioenergy, despite being a frequently inclusion in CBE definitions. As such, bioenergy is part of the lowest valorisation tier, while raw manure fertilisation would be assigned to the "Bulk Chemical and Materials" tier. Therefore, manure-based bioenergy technologies resemble a decrease in circularity.

However, due to the cascading effect, bioenergy is not mutually exclusive but is an additional valorisation step of any CBE pathway. Instead of solely using the biomass for its conventional usages, an additional bioenergy valorisation steps is included.

As for any CE scenarios, there is no scientific consensus about the sustainability of any individual CBE pathway. Therefore, individual sustainability assessments of CBE pathway are crucial (Carus and Dammer 2018)

2.1.3 Electronic Marketplace

No uniform concept or commonly agreed definition of EM exists (Stockdale and Standing 2004). However, many definitions include the provision of a digital market for products or services and connecting multiple buyers and multiple sellers (Taghipour *et al.* 2021).

A common goal of EMs is to facilitate trading between market participants. The concrete functionality of an individual EM depends on the identified market (Stockdale and Standing 2004). Among others, an EM can allow for product listings, price and quality information exchange, and procurement (Taghipour *et al.* 2021).

EMs can reduce transaction costs and provide a convenient solution, as transactions can be arranged and conducted on the EM without no face-to-face interactions or off-platform (Stockdale and Standing 2004).

2.2 Biowaste and Waste-based Bioenergy Technologies

In this chapter, an overview of the state of research regarding suitable bioenergy pathways in Finland is provided. This is seen as representative of the Hämeenkyrö municipality.

For the purpose of this thesis, three types of biowaste are examined more closely. The first selected biowaste is cattle manure, as it is the most produced biowaste in Hämeenkyrö (SDU 2024). Beside other types of manure, grass clippings from nature conservation sites, and household food waste are the most abundant biowaste types (MTK 2025b).

Next, the current market situation of these biowaste types in Finland is examined. Also, two bioenergy technologies, which are potentially relevant to the Finish context, are examined.

2.2.1 Finnish Biowaste Pathways

On a national level, 92.8 % of manure is reported to not be valorised beyond field application (Bywater *et al.* 2025). Additionally, a goal has been set to triple the national biowaste treatment capacity until 2027 (Ministry of the Environment 2022). This gives insight into the underdeveloped state of this sector.

Generally, the market for digestate is generally still in development. Commonly, limited digestate disposal options are available, which hinders can hinder the biogas potential (Dahlin *et al.* 2015). Biowaste is found to have a low economic value. In some cases, transport costs drive buyers' transaction costs above their willingness to pay. This is problematic for further bioenergy development, as decision-making is profitability driven. This profitability problem can partially be attributed to an incomplete market for biowaste (Huttunen *et al.* 2014).

Therefore, development improvement of the biowaste market is possible. One possibility is the introduction of an EM. The following subchapter provides an introduction into EM, followed by an introduction into a recently established EM for biowaste in Finland.

CircularFinland

As part of the TREASoURcE project a Finnish biowaste EM named "CircularFinland" (Finnish: "KiertoaSuomesta") has been established in 2022. The stated goal is to help sellers and buyers to find biobased materials. The EM functions as a business-to-business listing platform (MTK 2025a).

An apparent target group are farmers, which are both sellers and buyers of biomass. At the time of this thesis, predominantly agricultural and silvicultural biowaste and biowaste related services are listed. Among them, potential bioenergy feedstocks and digestate are listed (MTK 2025a).

2.2.2 Biowaste Technologies

In this chapter, two potentially interesting biowaste derived bioenergy technologies are presented. Anaerobic digestion is selected, as it is generally considered an established and therefore realistic bioenergy pathway (Kougias and Angelidaki 2018). Additionally, the biomass gasification technology is selected to represent the technological potential of bioenergy.

2.2.2.1 Manure-based Biogas

Biogas refers to a mixture of gases derived from a process called anaerobic digestion (AD). In this process, a biological feedstock such as manure is digested with insufficient oxygen supply. As a result, methane and carbon dioxide are emitted, which is captured as biogas (Felton *et al.* 2014).

Technology

The manure-derive biogas offers a unique combination of municipal biowaste treatment and bioenergy generation (Huttunen *et al.* 2014). AD is considered a “flexible technology” (Bacenetti *et al.* 2016: p.683) regarding its range of potential feedstocks. Biowaste is found to be a more environmentally beneficial feedstock than, for example, purpose-grown energy crops (Bacenetti *et al.* 2016).

Alongside biogas, AD also produces digestate as a biological side stream. Depending on the utilized biogas feedstock, this digestate can be used as an organic fertilizer (Huttunen *et al.* 2014). In this case, digestate becomes a raw-material of another production process which is referred to as “open-loop-recycling” (Klöpffer and Grahl 2014).

The biogas yield heavily depends on the feedstock. Cattle manure is found to possess a low biogas potential. This causes uncertainty of the specific electricity and heat generation. To increase the biogas yield, co substrates can be added to the feedstock to improve the C / N ration of the feedstock. This technology is referred to as anaerobic co-digestion (Esteves *et al.* 2019).

Implementation in Finland

AD is a rather established and researched technology (El-Mashad and Zhang 2010). However, currently almost all Finish cattle manure is directly applied on agricultural fields, while only around 2.5 % are used for anaerobic digestion (Bywater *et al.* 2025).

In Finland, biogas plants utilize several types of biowaste. The exact biowaste used commonly depends on the plant operator and their primary goals. However, the most commonly used biowaste type is wastewater slurry. Agricultural residue have not been found to be used regularly as an AD feedstock (Huttunen *et al.* 2014).

Environmental Impacts

Manure-based bioenergy pathways are found to be environmentally beneficial to conventional pathways (Boulamanti *et al.* 2013). This makes waste-based bioenergy especially interesting as bioenergy from purpose-grown energy crops have been found to have higher environmental impacts (Boulamanti *et al.* 2013; O'Keeffe *et al.* 2016).

Biogas systems are decentralized and can have different primary goals. For example, biowaste treatment or energy production (Huttunen *et al.* 2014). Also, biomass production varies regionally depending on climate, soils, and farm management (O'Keeffe *et al.* 2016). Therefore, biogas systems differ regionally.

A review of biogas LCAs highlights the importance of including the regional context in assessing the sustainability of bioenergy production. For example, the additional transport emissions of biowaste in a manure-based biogas system can play an important role in the overall impacts (Bacenetti *et al.* 2016). Consequently, generic sustainability assessments can lead to faulty results. (Huttunen *et al.* 2014).

2.2.2.2 Biomass Gasification

The second technology of interest is biomass gasification. Biomass gasification is a thermochemical conversion of dried biomass to syngas with limited amount of air (Nguyen and Hermansen 2015). The produced syngas consists of a mixture of energic gases such as hydrogen, methane, as well as other gases such as carbon dioxide and carbon monoxide (Xue *et al.* 2014).

Biomass gasification is found to be an "economically and environmentally beneficial technology" (Lundgren *et al.* 2025: p.2). However, it is still regarded as a "less mature technology than anaerobic digestion" (Nguyen and Hermansen 2015: p.81), despite being long-discussed, for example in Stevens (2001).

Ptasinski (2008) identified that grass is a more suitable gasification feedstock than manure. Therefore, this thesis limits its investigation of the gasification technology to only grass gasification.

2.3 Introduction to LCA

The LCA methodology aims at quantifying the environmental burdens of a product system throughout its life cycle. LCA case studies can help in the identification of environmental improvement potential across a products life cycle. For example, this can be helpful to producers, consumers, and policymakers (DIN 2006).

The LCA methodology serves as groundwork of this thesis' methodology. The following subchapters introduce the traditional LCA methodology as well as two novel LCA approaches.

2.3.1 Phases of Traditional LCA

The traditional LCA methodology is structured in four phases: 1. Goal and Scope Definition, 2. Life Cycle Inventory Analysis, 3. Life Cycle Impact Assessment and 4. Life Cycle Interpretation (DIN 2006).

In this chapter, the standardized methodological steps of these four phases are characterized. Due to the limitations of this thesis, this overview focuses on the methodological aspects that are most relevant for this case study. In the following subchapter, key steps of the methodological phases are illustrated by an example case study in an agricultural context.

2.3.1.1 Goal and Scope Definition

The first phase of an LCA is an introductory phase (Frischknecht 2020). All methodological decisions in the later stages of the LCA are made in accordance with this goal and scope definitions (European Commission 2010) and can therefore be highly relevant to the assessment's results (Frischknecht 2020). Consequently, all decisions made should be justified and should be made transparent to the recipient (Frischknecht 2020).

The first procedural step of the goal and scope definition describes the goal of the assessment. Here, the intended application, the objective, and the target audience of the LCA are defined (DIN 2006). These terms are subsequently explained and illustrated by an example of an LCA of milk and a milk alternative.

The first part of defining the assessment's goal is to declare the intended application for the LCA. This may be for example to document the current environmental status of a product or service. Commonly, an LCA is also intended to support the decision-making process of a company or in policy decisions. Alternatively, an LCA may also be performed as a methodological research study with its intended application inside an academic discussion (European Commission 2010).

Secondly, while describing the objective of an LCA, details about the concrete background of the LCA should be disclosed (European Commission 2010). In the above example, an LCA about different milk products may serve the objective of offering detailed advice to a specific retail discounter in an upcoming marketing campaign or consult regional policymakers about the potential environmental outcomes of a specific policy.

Finally, the target audience is named. A milk case study may for example be targeted at milk consumers with no technical experience on the LCA methodology. Therefore, methodological decisions should be made so that LCA results are relevant to the individual consumer and easily comprehendible. Another example is LCA experts, who can understand complicated analyses and highly technical language.

After the goal definition, the scope definition step introduces the LCA's object(s) of interest in greater detail. As part of that, the concepts of a functional unit (FU), product system and system boundaries are explained.

The Functional Unit

The first step in the scope definition is to determine the quantitative functions of the objects of interest. Importantly, those determined functions must also refer to the quality of the product in question (Frischknecht 2020). In the given example, milk fulfils multiple functions for consumers, for example reducing the effective bitterness of a coffee or adding desirable flavour to a drink.

In a "comparative LCA" the environmental burdens of multiple objects of interest are compared. In this case, all objects of interest must share the identified key functions. In common LCA practice, minor functional differences between the products need to be ignored (Klöpffer and Grahl 2014). The compared objects should resemble relevant product alternatives and should include innovative production methods (Frischknecht 2020).

In the next step, all selected functions are quantified. These quantified functions are referred to as FU. They describe "'what', 'how much', 'how well', and 'for how long'" a function is fulfilled by the object of interest (European Commission 2010: p.60). The FUs must be clearly defined as they serve as the basis of the LCA (DIN 2006). Through the FU, products of different quality can be quantitatively compared (European Commission 2010).

This concept is illustrated in the given example: It is assumed that 1 g of sugar from milk and 1 g of sugar from soy milk have the same effect of lowering the bitterness of the coffee. Therefore, an adequate FU for both liquids may be: "1 g of sugar in a milk drink".

An LCA may employ multiple FUs (DIN 2006). Continuing the above example, it is assumed that 1 g of fat from cow's milk possesses the same taste quality as 1.2 g of fat from soy milk. Therefore, a second FU can be described as: "1 g of cow's milk fat equivalences".

Product System Model and System Boundaries

The second step in defining the scope of the LCA is to model the life cycle of the object of interest (Frischknecht 2020). Here, so-called product system models are created, which each consist of multiple so-called unit processes that each describe an event during the product's life cycle stages

(Klöpffer and Grahl 2014). A products life cycle can in principle be separated in the four stages: Raw material extraction, Production, Use and Disposal (Frischknecht 2020).

Unit processes are the smallest elements of a product system. They may represent, for example a milking, feeding, or transportation process. Unit processes contain in- and outputs, so-called flows. Each flow describes an amount of a material or energy and links to another unit processes or the ecosphere. Flows that link to the latter are also referred to as elementary flows (Klöpffer and Grahl 2014).

The full life cycle of most product systems is too complex to be fully broken down into unit processes. Therefore, a range of different system boundaries are defined, which limit the extent of the product model. A commonly cited system boundary approach is the “cradle-to-grave” approach. In this approach all processes from resource extractions through production, use, and ultimately disposal processes are included (Klöpffer and Grahl 2014)

Technical system boundaries are also essential to limit the extent of an LCA in practical work (Klöpffer and Grahl 2014). These boundaries determine the minimal size of a flow to be included in the product model. According to Frischknecht (2020) a flow with a negligible mass and energy contribution shall be excluded from its unit process. This threshold is commonly set at 1 % of the mass of any process. However, the sum of the flows affected by this threshold may not exceed 5 % of the process (Frischknecht 2020). However, if it is otherwise known that a flow or process has a high environmental burden, they need be included in the product model (Klöpffer and Grahl 2014).

In addition to the above mentioned system boundaries, the product model is also specified geographically and temporally (Frischknecht 2020). For example, the study subject is limited to milk-drinks produced in Germany during the year 2025.

It should be highlighted that processes and environmental impacts occurring outside these geographic or temporal boundaries must not be excluded from the assessment. Instead, these boundaries are set to specify the product system of interest more precisely. Also, the technology level of the processes in the product system is specified, ranging from best-available technology to phasing-out technology (Frischknecht 2020).

As part of the scope definition, it is defined what kinds of environmental impacts (impact categories) are assessed in the LCA. These impact categories shall cover all environmental burdens which are deemed to likely be influenced by the product systems (European Commission 2010).

If the LCA is a comparative LCA, in the final procedural step the compared products are stated (European Commission 2010). Comparative LCA should include innovative production varieties (Frischknecht 2020).

It should be highlighted that all methodological steps of an LCA are iterative. Therefore, methodological decisions made in previous LCA phases should be revised when new insights are available (DIN 2006).

2.3.1.2 Life Cycle Inventory

In the second phase of the LCA, numerical data for each unit process is collected and aggregated in a so-called life cycle inventory (LCI). For this purpose, the activity of each process is described as a linear mathematical formula.

An LCI contains all interactions between the anthropological product system inside the system boundaries and the environment. Those are also referred to as the technosphere and ecosphere. The result of an LCI is a list of all emissions to the ecosphere including the atmosphere, water bodies and soils as well as resource usage (Frischknecht 2020).

European Commission (2010) argue that the differentiation between these two spheres in an agricultural context can be especially challenging. For example, the cultivation of an agricultural plot is a clear anthropological interference with the ecosphere and should therefore be represented in the LCI. However, the boundary between those two spheres appears to be practically fluent. (European Commission 2010). For example, it may be unclear what share of soil erosion, and its entailing carbon loss, should be attributed to the cultivating process.

During the data collection, effort should be made to confirm the validity of the collected data. DIN (2006) recommends checking for shortcomings in the laws of conservation of energy and mass in each unit process. However, as Klöpffer and Grahl (2014) point out, the laws of conservation of mass and energy may not be fulfilled precisely, due to the system boundaries set in the previous phase.

Attributional vs. Consequential LCI approach

Depending on the goal of the analysis, an LCI can follow an attributional or a consequential modelling approach. An attributional LCI contains all flows that are relevant to the life cycle of the defined product system. By contrast, a consequential LCI includes all environmental impacts that are the result of the defined product system instead of another product system (Frischknecht 2020). Therefore, a consequential LCI additionally includes the LCI of those substituted product systems and therefore requires additional assumptions about the surrounding economy (Frischknecht 2020).

A consequential LCI should also include other processes from the technosphere. (European Commission 2010). For example, it may be reasonable to assume that the production of soy milk reduces the production of cow's milk. However, European Commission (2010) also mention that in LCA practice this step is commonly simplified.

Data Provision

To achieve the highest data accuracy, data of the foreground should be collected on site of each unit process. However, in LCA practice, a compromise between data accuracy and temporal and financial limitations of the LCA must be struck. This is especially relevant as data acquisition is frequently referred to as one of the most time intensive steps of an LCA (European Commission 2010).

In practical LCA work, the usage of estimates and assumptions is often necessary (Frischknecht 2020). Frischknecht (2020) stresses that LCI data should not solely originate from other LCA studies, as possible flaws in the data would inevitably be inherited.

To compile a comprehensive LCI, generic background databases that contain data from thousands of processes are commonly deployed. Extensive background databases are especially needed when the supply chain of a product is inconsistent over time or is not traceable, such as electricity (Klöpffer and Grahl 2014).

In many cases, the environmental burden of an emission heavily depends on the location that it occurs in. This is very typical for emission in an agricultural context, where for example the freshwater usage in temperate cause lower environmental burdens than in arid regions. The location of each unit process should therefore also be contained in the foreground model (Frischknecht 2020).

Allocation Rules

Unit processes that have multiple co-products are also called multiproduct processes. Those are especially common in agricultural processes, for example, a farm raising dual-purpose cows produces e.g. milk, meat, and horns.

However, this common circumstance poses a methodological challenge to conducting an LCI. Finding a science-based and "fair" allocation rule of environmental burdens between co-products is not trivial (Klöpffer and Grahl 2014). In the methodology of LCA a hierarchy of allocation rules has been established (Frischknecht 2020). In this chapter, the first four of these rules are presented in hierarchical order.

Before that, however, it should be differentiated that co-products without an economic value are not considered a co-product, but a waste product. Waste products are not attributed any environmental burden, but only to the purposed outputs (Klöpffer and Grahl 2014).

The first allocation rule is to check for possible avoidance of an allocation. In the subdivision approach, each process that is exclusively related to a single co-product is assigned to that co-product

(European Commission 2010). In the example of dual-purpose cow rearing, any process that is only relevant to the processing of milk should not be included in the product system of beef. This approach is plausible for example for processes such as milking and milk transport.

The second allocation rule is named “system expansion” (DIN 2006). Here, an alternative process from the surrounding economy that provides one of the co-products is identified. In the system expansion logic, the production of one unit of the co-product effectively avoids the production of one unit of the compared product. The LCI is then credited with avoiding the production of that output. The LCI of that avoided product is subtracted from the multiproduct process’s LCI. This system expansion approach is complete when only one output from the multiproduct process remains (Frischknecht 2020).

Building on the previous examples, cows in southern Germany are commonly dual-purpose breeds with at least their milk as well as their meat being marketed (Tergast *et al.* 2025). The system expansion approach requires a process from the surrounding economy that only produces, e.g. beef. In the given example, it is within reason to equate the function and quality of beef from this multi-product product system to beef from a single product system, where only beef is produced. The LCI of that avoided product is then subtracted from the LCI of the dual-purpose cow farm.

Klöpffer and Grahl (2014) highlight that system expansion requires additional assumptions and therefore increases the data requirements of the LCI.

The third form of allocation is based on the physical causation between the co-products (Klöpffer and Grahl 2014). This allocation approach is only eligible if for co-products that share a common physical property, which is also decisive for their economic value (DIN 2006).

A common example is the case of a crude oil refinery with a multitude of fossil fuel outputs. The common property of those co-products can be set as the energetic potential of the fuel outputs. In agriculture, this method is rare, as outputs commonly vary widely in their intended uses and quality (Jungbluth 2023).

The fourth allocation rule is based on the economic value of the co-products. The economic value of the products is determined for one production cycle, often one year (Klöpffer and Grahl 2014). Due to the practical constraints and necessary additional assumptions of the previous approaches, economic allocation is very relevant in practical LCA work (Jungbluth 2023).

End-of-Life phase

The last phase in a product’s life cycle is called end-of-life (EoL) phase, which is a common inclusion in most LCA case studies, depending on the defined system boundaries. It commonly deals with waste treatment processes such as landfilling, incineration or recycling (Klöpffer and Grahl 2014).

In the case of recycling, a methodological decision must be made about the allocation of the secondary raw material from recycling between their first and second life cycles. For example, in the commonly applied “Cut-Off rule” burdens of the first life cycle, e.g. resource extraction, are not allocated to the material’s secondary life cycle (Klöpffer and Grahl 2014). Consequently, the secondary material is not allocated any upstream emissions.

By contrast, the avoided burden approach grants credit for avoiding primary resource extraction to the recycled product in the secondary life cycle (Frischknecht 2020).

2.3.1.3 Life Cycle Impact Assessment

During the life cycle impact assessment (LCIA), the potential environmental impacts of the product system are quantified. Those environmental impacts cover different category endpoints such as to the natural environment, human health and resource depletion are calculated (European Commission 2010).

The endpoint categories are divided into different impact categories such as climate change, or freshwater eutrophication (Klöpffer and Grahl 2014). As mentioned in Chapter 2.3.1.1, the selection of relevant impact categories is already established during the Goal and Scope phase of the LCA. Due to LCA being an iterative method, the selection can still be adapted to best represent the intentions of the LCA (Frischknecht 2020).

In the following subchapter, the procedural steps of LCIA are illustrated by the common and intuitive example of climate change. Due to the limitations of this thesis, this chapter focuses on the LCIA steps that are essential to the LCIAs functionality and are relevant for this case study.

To improve readability of all subsequent chapters, “potential impacts” are further referred to as “impacts”.

Procedural steps

In the first step of LCIA, all elementary flows from the LCI are classified based on their relevant to an impact category of interest. For example any greenhouse gas (GHG) is classified to increase the global infrared radiative forcing (Frischknecht 2020).

Secondly, during the characterization step, all classified elementary flows are assigned a respective so-called characterisation factor (CF). In the given example, each elementary flow is assigned its quantitative infrared radiative forcing impact in CO_2-eq (Klöpffer and Grahl 2014). For example, Andreasi Bassi *et al.* (2023) recommend using a CF of 1 for fossil carbon dioxide, and a CF of 29.8 for fossil methane. In common LCA practice, these CFs are assigned by a generic external evaluation method, which are followingly also referred to as LCIA methods (Frischknecht 2020).

To calculate the LCIA for an impact category, the quantity of each classified flow is multiplied by its respective CF. Any elementary flow is calculated to either cause a net burden or generate a net benefit, also referred to as a net credit. This procedure of classification and characterization is then repeated for all impact categories of interest (Frischknecht 2020).

Commonly, elementary flows are relevant to multiple impact categories. Therefore, the amount of LCIA calculations required in an LCA case study requires computational assistance in the form of dedicated LCA software (Klöpffer and Grahl 2014).

Identifying the significant issues from the LCIA results is not trivial, as each impact category implays a different impact indicator. Therefore an optional normalisation step can be conducted in which the most affected impact categories are asserted (Frischknecht 2020).

For that purpose, the LCIA results of each impact category are divided by a normalisation factor (Klöpffer and Grahl 2014). Previously, various sets of normalisation factors have been published based on different reference values. For example, Frischknecht (2020) cites the total regional impacts per-person, or the impacts resulting from an alternative product system, among others.

2.3.1.4 Interpretation

In the fourth and final phase of an LCA, a reflection on the results of the LCI and LCIA are formulated (DIN 2006). Those should include insights into the achieved accuracy of the results, how well the analysis represents the goal of the analysis, and the limitations of the assumptions made during the other LCA phases (European Commission 2010). As part of the iterative nature of LCA, these reflections should be used to further improve the product system to better represent the goal and scope of the case study.

In this LCA phase the LCA methodology grants more liberties to the LCA practitioner than in the previous phases. The reviewed secondary literature offers more diverse guidance and emphasizes different interpretation aspects. For example, Laurent *et al.* (2020) suggests a different order of evaluation analyses as DIN (2006).

First, based on the defined goal and scope of the case study, the significant findings are identified and structured. Also, as part of the evaluation, the LCI and LCIA results should be compared with results of similar case studies. Then the robustness of these results is assessed via different data quality analyses. For example, the sensitivity of the LCIA results to data and methodological choices are tested (European Commission 2010). The following subchapter describes the procedures of a sensitivity analysis in greater detail.

Sensitivity Analysis

A sensitivity analysis or sensitivity check is "probably the most frequently applied quantitative technique" (Klöpffer and Grahl 2014: p.333) to identify the uncertainty in the LCIA results to the data quality or the methodological decisions made (Klöpffer and Grahl 2014).

A sensitivity analysis is typically performed for individual in- or output flows and activity data that has been found to contain high uncertainty and/or is highly significant to the LCIA results (Frischknecht 2020). When a choice between allocation approaches has been made during the LCI phase, this evaluation step even becomes mandatory, according to DIN (2006).

Sensitivity analyses are conducted by comparing different analysis scenarios. Each scenario features a change of one or more LCI variables or methodological decisions. The assessment is then re-run and the variation in the results between scenarios is observed (Frischknecht 2020).

Due to the iterative approach of the LCA methodology, findings made during this step should be used to reevaluate the significant issues identified in the previous step. Data that is identified as especially sensitive to the LCIA results, the data quality shall be improved where feasible (European Commission 2010).

2.3.2 Novel LCA Approaches

The research questions of this case study exceed the scope of the above-described traditional LCA methodology. First, to assess the regional impacts of a product system, a more context-dependent sustainability assessment is required. Secondly, traditional LCA is not equipped to assess the impacts over a larger time frame. Therefore, this chapter introduces two extensions to the traditional LCA methodology: context-based LCA and prospective LCA (pLCA). In the following subchapter, those approaches are briefly described.

2.3.2.1 Context-Based LCA

As mentioned in the chapter above, the traditional LCA methodology already recommends the inclusion of the regional context during the LCAs phases. Further regionalisation can for example be achieved through geographically accurate LCI data or regionalised CF modelling (Frischknecht 2020). This is relevant in an agricultural context, when, for example, the regional damage of additional nitrate pollution depends on the pre-existing nitrate pollution in groundwater.

However, in practical LCA work, specific regional LCI data is often unavailable, while on-site data collection often proves to be unfeasible. Therefore, LCA case studies frequently transfer regional data from other regional LCA studies and are therefore agnostic to characteristics of their specific study region (European Commission 2010).

The context-based analysis stage of this thesis utilizes the “within region” approach, where environmental burdens are caused outside the region are not the focus. This is an interesting approach for bioenergy systems, if production and usage is assumed to occur in same region or when bioenergy is traded supra-regionally. Based on this approach, for example, avoided fertilizer production from outside of the region of interest is excluded (O’Keeffe *et al.* 2016).

For this thesis, the regional context of the LCA is additionally included through an absolute environmental sustainability assessment (AESA) approach (Guinée *et al.* 2022). This approach has previously been used to assess and compare the absolute sustainability of e.g. countries or regions (Cole *et al.* 2014). Subsequently, the purpose and functionality of the AESA methodology is explained.

AESA

Traditional LCA methodology is capable of assessing the relative sustainability of a product system compared to another product system (Bjørn *et al.* 2020). However, as detailed in Chapter 2.1, it has become apparent that the global earth system also has absolute emission limits before critical tipping points to the earth system functionalities are reached.

In an AESA, the calculated LCIA results are compared to an absolute sustainability benchmark (Bjørn *et al.* 2020). Various forms of quantifying the safe operating spaces for LCA-purposes have been documented in the scientific literature. For example, Bjørn and Hauschild (2015) introduced the “carrying capacities” based on the PB framework. These quantify maximal indicator results of different impact categories, which can then be used to normalize the LCIA results of any LCA case study to quantify its depletion of the safe operating spaces (Bjørn and Hauschild 2015).

Besides the PB framework, another prominent example of an absolute sustainability benchmark is the global goal of limit the average global temperature increase to 1.5 degree. Hence, Steffen et al. (2015) calculated the amount of GHG emissions permitted to meet this goal and transferred them to carrying capacities for the climate change impact category.

Depending on the goal and scope of an LCA, the global carrying capacities are scaled down to represent the study objects’ share of the safe operating spaces. For that purpose, Bjørn *et al.* (2020) identified multiple sharing principles. For example, the carrying capacities of a region can be shared per-person or share per-land area.

Disaggregation of carrying capacities between the different economic sectors includes additional methodology difficulty and uncertainty (Müller *et al.* 2025), which are beyond of the limitations of this thesis.

2.3.2.2 Prospective LCA

The pLCA is a novel LCA approach that assesses the environmental impacts of a product system into the future. This requires additional assumptions about the product system and the surrounding economy (Arvidsson *et al.* 2018).

These assumptions can be used in a future scenario analysis. In such a future scenario, each scenario describes a consistent global development path that leads to a distinct future situation (Bisinella *et al.* 2021). This approach is described to be especially useful in decision making support (Godet 2000).

The REMIND model by Baumstark *et al.* (2021) provides various future scenarios across different time horizons, so-called integrated assessment model (IAM) scenarios. The assumptions of each IAM scenario can be divided into two major modules:

In the first module, different assumptions about the international "long-term macroeconomic developments" (Baumstark *et al.* 2021: p.6572) are made. Those include assumptions regarding national GDP, education, and demographics, among others (O'Neill *et al.* 2014). These modules are therefore referred to as shared socio-economic pathways (SSP).

The second module makes assumptions about the trajectory of GHG emissions, referred to as the representative concentration pathway. Each representative concentration pathway is an emissions scenario based on the degree to which global GHG emissions target are assumed to be achieved by the year 2100 (Sacchi *et al.* 2022).

Based on these two modules, each IAM scenario makes the most relevant changes to the national production and production efficiencies of electricity, steel and cement as well as road transport (Sacchi *et al.* 2022).

Arvidsson *et al.* (2018) stress the importance of "avoiding temporal mismatch between the foreground and background systems" (Arvidsson *et al.* 2018: p.1292). Therefore, pLCA studies should employ assumptions about both the foreground and background model of the LCI (Arvidsson *et al.* 2018).

2.4 Hypotheses

Based on the previous chapters, the following hypotheses about the CBE scenarios of interest are made:

i. Treating biowaste in valorisation pathways leads to lower environmental impacts than treatment in conventional pathways (Huttunen *et al.* 2014) (Esteves *et al.* 2019).

ii. A biogas feedstock of cattle manure with co-substrates like food waste and grass clippings further increases the environmental credit of anaerobic digestion (Esteves *et al.* 2019).

iii. Thus, the implementation of a biogas plant in Hämeenkyrö reduces the municipality's depletion of its carrying capacities.

iv. Transportation plays an important role in the valorisation impacts (Esteves *et al.* 2019; Bacenetti *et al.* 2016).

v. Over the next decades, the environmental performance of the valorisation pathways and scenarios worsens due to increased share of regenerative sources in the substituted energy (Baumstark *et al.* 2021).

3 Methodology

This case study investigates multiple hypotheses regarding the environmental impacts of different CBE scenarios in Hämeenkyrö. For this purpose, this thesis introduces a dedicated SAF.

The SAF addresses the different hypotheses via multiple analysis stages. The first two analysis stages are based on the traditional LCA methodology described in Chapter 2.3.1, which is simplified to remain inside of the limitations of this thesis. The third and fourth analysis stage additionally include the novel LCA approaches introduced in Chapter 2.3.2.

Following, the methodological steps of the traditional LCA analysis stages are presented. After that the procedure of the novel LCA is explained. A combined interpretation step for all SAF analysis stages is performed as part of the discussion in Chapter 5.

3.1 Traditional Analysis Stages

In the first analysis stage, an overview of the conventional and valorisation pathway impacts is given in an attributional analysis. For that purpose, the pathways are compared for FU1: 1 tonne of biowaste treatment.

In the second analysis stage, the valorisation pathways are compared in a consequential analysis. Here, the same FU as in the previous analysis is used. However, each pathway is assumed to avoid the conventional pathways of their respective biowaste feedstock. Also, the outputs of the product system are assumed to substitute a marginal product from the surrounding economy with full elasticity. However, the system boundaries of this analysis include the avoided processes from the conventional biowaste treatment pathways.

3.1.1 Goal and Scope

The results of this thesis are intended to advise municipal policymakers of Hämeenkyrö in their decision-making process. Additionally, this case study is aimed at laying the groundwork for future replication for other municipalities. This target group is assumed to be knowledgeable about the LCA methodology as described in Chapter 2.3.

Functional Unit

As mentioned above, the objective of this thesis is to assess the environmental impact of different biowaste treatment pathways. The common function of all pathways is their occupancy of municipal biowaste treatment capacity. Therefore, the pathways are compared for 1 tonne of biowaste treatment. A second FU is defined that deals with the combined biowaste treatment of the

Hämeenkyrö municipality. Both FUs feature the secondary functions of electricity, heat, and organic fertilizer generation.

Product System Model and System Boundaries

Here, the product systems of all pathways are described. All product systems are based on the "cradle-to grave" principle. Based on O'Keeffe *et al.* (2016), three life cycle stages of biowaste treatment are identified as: Biowaste recovery, production, and fertilization.

The conventional pathways are subsequently referred to as conventional manure management (Conv.MM), household food waste incineration (FW-Incin) and decay of grass clippings on nature conservation sites (Grass-Decay). Each conventional pathway is based on the assumed currently predominant biowaste treatment method in Hämeenkyrö (MTK 2025b).

The following Figure 4 provides a simplified overview of the life cycle stages in the conventional biowaste treatment pathways of three types of biowaste:

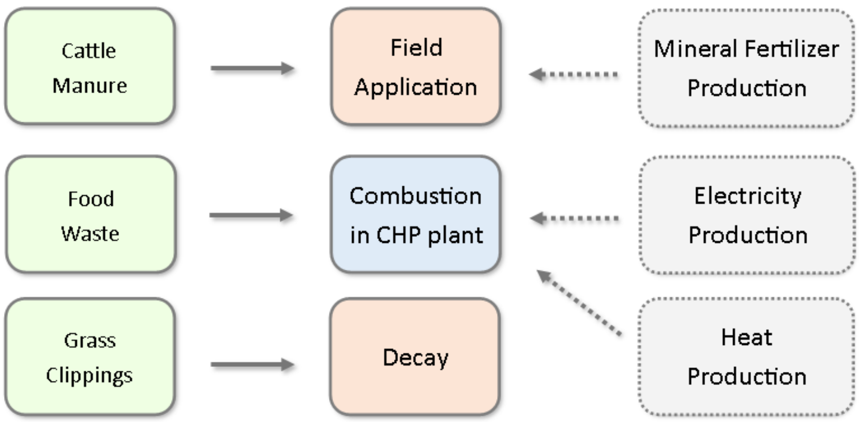

Figure 4: Simplified overview of the life cycle stages of the conventional pathways. Colours represent life-cycle stages: green = biowaste recovery; blue = production; red = fertilization; dotted arrows = substitution.
Source: Own design.

The overview reveals that the conventional pathways contain few processes and life cycle stages. Also, only a limited connections to other product systems in form of substituted processes exist.

In comparison to that the two bioenergy technologies are considered as alternative valorisation pathways. As highlighted above, biogas systems depend on the regional context, such as their primary goal. Therefore, three biogas feedstocks are considered: One pathway deals with manure mono-digestion without any co-substrates, while two pathways deal with the mixture of manure feedstock with food waste or grass, respectively. These are named: Mono-digestion (Mono-AD), manure and household food waste co-digestion (FW-CoAD), and manure and grass co-digestion (Grass-CoAD).

As mentioned above, comparative LCA should also include innovative production methods. Therefore, the valorisation pathways also features a grass gasification pathway (Grass-Gasf).

Each pathway consists of around six to eight processes that describe the biowaste primary use, as well as a side stream life cycle stage. The following figure provides a simplified overview of all valorisation pathways:

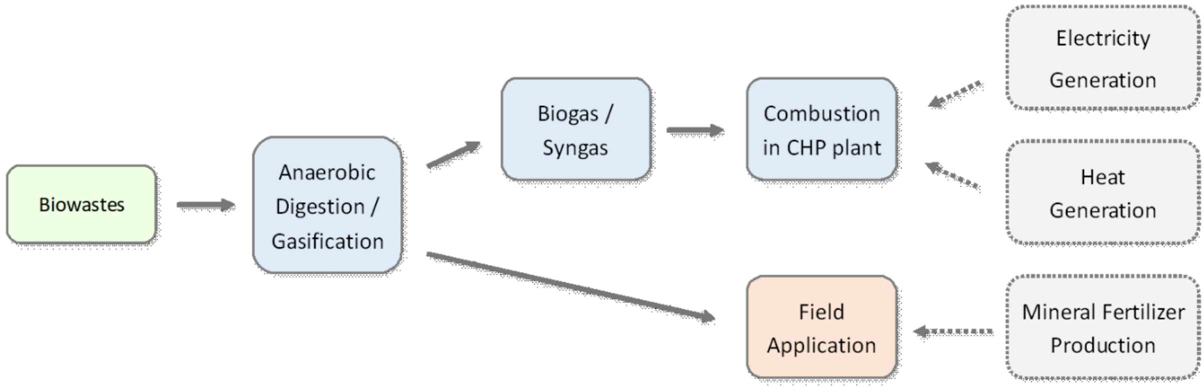

Figure 5: Simplified overview of the life cycle stages of the valorisation pathways. Colours represent life-cycle stages: green = biowaste recovery; blue = production; red = use stage; dotted arrow = substitution.
Source: Own design.

The valorisation pathways introduce new processes to the life cycle stages of the biowaste. For example, food waste hygienisation, AD / gasification and digestate / biochar field application. Like Huttunen *et al.* (2014), this thesis assumes the usage of digestate for field applications is permitted. Therefore, each pathway includes all three substituted processes.

Regarding the geographic, temporal and technological system boundaries: This case study only deals with biowaste that currently originates from the Hämeenkyrö municipality. All processes are assumed to be of the average Finnish technological standard.

Impact Categories

The scope of this thesis confines the SAF to a selection of five focus impact categories. These are selected based on the most commonly assessed impact categories in relevant literature reviews of LCAs of manure-based biogas (Huttunen *et al.* 2014; Bacenetti *et al.* 2016; Esteves *et al.* 2019). Additionally, the freshwater ecotoxicity impact category is included as agricultural products are known to cause significant ecotoxicity impacts (Jungbluth 2023).

Therefore, this thesis focuses on the following five impact categories: acidification, climate change, freshwater ecotoxicity, freshwater eutrophication, and marine eutrophication. Exploratory findings from additional impact categories are discussed in Chapter 5.3.3.

3.1.2 Life Cycle Inventory

In this chapter, the procedure of compiling the LCI of the product systems is described. First, the compilation steps of the foreground LCI data are explained. Then, the implementation of the EM is described. Finally, the selection of the background LCI database is justified.

Foreground Model

LCI data acquisition is "one of the most complex phases of LCA" (Klöpffer and Grahl 2014: p.67). Due to the limitations of this thesis, it is determined that the foreground LCI of the SAF is based on previously compiled LCI data from similar LCA case studies. This decision is made despite the methodological reservations mentioned in Chapter 2.3.1.2 regarding LCI data transfer.

These LCI data source studies are selected based on similar objects of interest, goal and scope and high LCI comprehensiveness. The case studies are obtained using Google Scholar and JSTOR. The snowballing approach is employed to extent the research to additional data sources.

The foreground LCI data sources of the pathways of interest are presented in the following table:

Table 1: Overview of the foreground LCI data source studies.

Conventional Pathway	LCI Data Source Study	Study Region
Conv. MM	Pehme (2017)	Estonia
FW-Incin	Hamelin (2014)	Denmark
Grass Decay	Pehme (2017)	Estonia

Valorisation Pathway	LCI Data Source Study	Study Region
Mono-AD	Pehme (2017)	Estonia
FW Co-AD	own calculations, based on Pehme (2017) and Hamelin (2014)	Estonia, Denmark
Grass Co-AD	Pehme (2017)	Estonia
Grass-Gasf	Nguyen and Hermansen (2015)	Denmark

Source: Own data and design.

In the scope of this thesis, no suitable LCI data for the FW-CoAD pathway is found. Therefore, the LCI of this pathway is compiled for this theses and based on the biowaste characteristics and formulas of the "Household biowaste scenario" provided by Hamelin *et al.* (2014). To maintain consistency with the other two pathways, the share of food waste in the biogas feedstock is based on Pehme *et al.* (2017).

For the LCI of the Grass-CoAD pathway, the scenario "Manure co-digestion with natural gras" from Pehme *et al.* (2017) is used.

All LCI data source studies of the foreground model have been conducted for Estonian or Danish conditions. Due to the geographic proximity of these countries to Finland, it is assumed that the foreground models are similar.

The collected LCI data is adjusted to correspond to the FUs of this study as well as to correspond to the mean annual temperature of Hämeenkyrö. Merkel (2025) observed a mean annual temperature of 5.1 °C for the nearby city of Tampere. This temperature is assumed to also be the mean annual temperature of Hämeenkyrö.

A complete overview of the LCI inventory of all modelled foreground processes is displayed in II. Table 2 presents key inventory results of the compiled LCI for all pathways on a process level. All values are adjusted to FU1: 1 tonne of biowaste treatment and rounded to three significant digits. Any variable that is not contained in the respective pathway is indicated by a hyphen.

Table 2: Overview of key variables in the attributional LCIs. FM = Fresh matter, Nm ³ = normal cubic meter.

		Conventional Pathways:			Valorisation Pathways:			
Flow	Unit / FU	Conv. MM	FW Incin	Grass Decay	Mono-AD	FW-CoAD	Grass-CoAD	Grass-Gasf
Cattle Manure in Feedstock	kg (FM)	1000	0	0	1000	748	751	0
Co-Substrate in Feedstock	kg (FM)	0	1000	1000	0	252	249	1000
Biogas Yield (65% CH_4)	Nm^3	-	-	-	35	59.4	53.9	-
Syngas Yield (unspecified)	MJ	-	-	-	-	-	-	5370
Net Electricity Yield	MJ	-	74	-	302	473	465	1610
Net Heat Yield	MJ	-	0.21	-	155	353	309	1720

Source: Own data and design.

All analysis stages assume that the implementation of any pathway neither impacts the amount of municipally produced biowaste nor changes the demand in the pathways' output. The plausibility of these key inventory results is discussed in Chapter iii.

Background Model

As the background LCI database of this case study the Ecoinvent LCI database is chosen, as it is one of the most used background databases (Weidema *et al.* 2013). The Ecoinvent database utilizes the common "Cut-Off" EoL rule regarding recyclable products such as biowaste. Therefore, biowaste is not allocated any environmental from their previous life cycle (Klöpffer and Grahl 2014).

For the purpose of this thesis, the Ecoinvent database has been modified by the premise database modification tool (Sacchi *et al.* 2022). Among others, this modification introduced different time frames of the IAM scenarios to the background database. The premise tool has been selected, as it is said to allow for the most developed pLCA background database (Bruhn *et al.* 2023). For consistency reasons, this premise database is used for all analysis stages of the SAF.

Some elementary flows contained in the LCI data sources are found to not be available in the premise database, such as "Ammonia to river". In these cases, an analogous elementary flow is put in place.

EM Integration

As detailed above the implementation of the EM is seen as a prerequisite for the establishment of the valorisation pathways in Hämeenkyrö. However, it is further assumed that the EM uptake improves the efficiency of the biowaste market and therefore decreases the required transport distances of the valorisation pathways.

In the scope of this thesis, no literature reference is found that quantifies the plausible reduction of transport distances due to the establishment of an EM. Therefore, the following assumptions are made:

The initial adoption of the EM by market participants has been found to be timid (Ngo 2023). It is therefore assumed that average transport volume is reduced by 5% in the present time frame. For the prospective analyses, it is assumed that by 2050 the EM has gained increased acceptance among buyers and sellers and therefore reduce the average transport volume required by 20 %.

3.1.3 Life Cycle Impact Assessment

The SAF employs the environmental footprint LCIA method (EF) in its version 3.1. The EF method is widely used in LCA practice and recommended by European Commission (2021).

The EF method contains a total of 16 impact categories. Among them, the five focus impact categories determined in Chapter 3.1 are included (Andreasi Bassi *et al.* 2023). This method is found to be the only comprehensive LCIA method that is compatible with the premise LCI background database.

All LCIA calculations are performed in the open source LCIA software openLCA (v2.5) (GreenDelta GmbH 2025).

3.1.4 Interpretation

To identify the most significant processes and elementary flows, each analysis stage features a hotspot analysis. These are conducted following the recommendations detailed in European Commission (2021). This methodology is selected as it is found to be the only feasible quantitative methodology highlighted in the reviewed literature on LCA interpretation.

According to European Commission (2021), a "most-relevant" elementary flow is defined as each of the biggest elementary flows that contribute to a cumulative absolute sum bigger than 80% of the impacts of a most-relevant process.

Due to the limitations of this thesis, the LCIA results are not disintegrated sufficiently to confidently identify the linkage between processes and elementary flow. Therefore, the most-relevant elementary flows across all processes of each pathway are pointed out.

This case study does not feature a dedicated interpretation chapter. Instead, the identification of significant issues is performed in the results chapter, while the LCIA evaluation steps mentioned in 2.3.1.4 are performed as part of this thesis' discussion.

3.2 Novel Analysis Stages

This chapter describes the adaption of the traditional LCA methodology for the third and fourth SAF analysis stages using the novel LCA approaches.

3.2.1 Context Based Analysis

In the third analysis stage of the SAF the impacts of four municipal development scenarios are assessed. The FU2 is used, which considers the entire biowaste treatment of all three types of biowaste on a municipal level.

The above-mentioned "within region" approach is used. The electricity, heat, and mineral fertilizer outputs are seen as interregional product, which are therefore excluded from this analysis stage.

In the first conventional scenario (Conv.Scenario), all biowaste is treated in their respective conventional pathways. This Conv.Scenario is compared to three biogas scenarios:

Communications with experts from MTK (2025b) indicated that the timely construction of a medium-scale biogas plant with an annual biowaste treatment capacity of 20,000 tonnes in Hämeenkyrö is realistic. In each biogas scenario, this biogas feedstock capacity is utilized exclusively by either the Mono-AD, FW-CoAD or Grass-CoAD pathway. Remaining biowaste is treated by their conventional pathways.

The development scenarios employ specific biowaste data from the Hämeenkyrö municipality from a Material Flow Assessment (MFA) (SDU 2024). This MFA comprises public data from the "biomass atlas" (LUKE 2020) with survey data about produced biowaste among farmers that cultivate fields in Hämeenkyrö. SDU (2024) acknowledged that the MFA survey data on municipal biowaste of Hämeenkyrö accounts for some biowaste produced outside the Hämeenkyrö municipality.

The MFA showed that the annual biowaste production amounts to 71,040 tonnes of cattle manure, 22,106 tonnes of food waste and 15,874 tonnes of grass clippings from nature conservation sites (SDU 2024).

AESA

In this analysis, the safe operating spaces described by the PB framework are employed as an absolute sustainability benchmark. For that purpose, municipal carrying capacities are calculated.

Sala *et al.* (2020) conveyed the global "safe" emissions of the PB framework into annual global impact limits of each impact category within the EF method. In the scope of this thesis, the global emission limits provided by Sala *et al.* (2020) are found to be the only established global emission limits compatible with the EF method.

First, the municipal carrying capacities are calculated. For that purpose, the global emission limits calculated by Sala *et al.* (2020) are scaled to the municipal level.

As mentioned above, multiple sharing principles have previously been utilized (Bjørn *et al.* 2020). For this regional assessment, the sharing principle is based on the geographic characteristics of the Hämeenkyrö municipality. For this context-based analysis, two sharing principles are tested:

In the first approach, the global impact limits are shared by the "Equal per capita" sharing principle (Bjørn *et al.* 2020). For that purpose, the safe global emission budget is divided by the global population and shared and multiplied by the inhabitant of Hämeenkyrö. The global population is assumed to be 7,856,138,790 (WBG 2025). A recent estimate of the population of the Hämeenkyrö municipality totals 10,339 inhabitants (Statistics Finland 2024). Consequently, the sharing factor of the carrying capacities is calculated to be $1.32 * 10^{-6}$.

In the second approach, the global impact limits are divided by the global land area and multiplied by the land area of Hämeenkyrö. This sharing principle has been referred to as the "Land area" sharing principle (Bjørn *et al.* 2020). The global land area is found to amount to 140.944.693,9 km^2 (FAO 2022). The land area of the Hämeenkyrö municipality is found to be 463.9 km^2 (NLS 2017). The resulting sharing factor is $3.29 * 10^{-6}$.

Finally, the global carrying capacities are now divided by the either sharing factor, to calculate the municipal carrying capacities per-capita or per-land area. These serve as normalisation factors of the LCIA results of the development scenarios and are presented as part of this thesis' results in chapter 4.3.

3.2.2 Prospective Analysis

As the fourth element of the SAF, the impacts of the pathways and development scenarios from the consequential and the context-based analysis stages are calculated for future time frames. For that purpose, each pLCA utilizes the FU and system boundaries of the respective underlying analysis stage.

In the scope of this thesis is only feasible to examine one IAM scenarios. As the first module the SSP2 scenario is selected as it represents a “Middle-of-the-Road” (Sacchi *et al.* 2022: p.3) scenario which is selected as it is considered to be more likely than any other more extreme SSP scenario.

As the second module, it is assumed that the energy investments are implemented according to national policies (National Policies Implemented - NPI). This module assumes that globally all countries implement their current national climate policies (Baumstark *et al.* 2021). Therefore, the share of regenerative energy in the national energy mixes are assumed to increase. This climate policy module is selected because it represents the status-quo regarding the global climate change mitigation ambitions and therefore is seen as most likely.

For this thesis, the IAM scenario of SSP2_NPi scenario is selected. Due to the limitations of this thesis, no technological efficiency increases can be implemented to the foreground model.

4 Results

In this chapter, the results of the SAF are presented and significant issues identified via the hotspot analyses. According to European Commission (2021) all most-relevant processes and elementary flows identified by the hotspot analyses are stated. The most-relevant processes and elementary flows are presented in descending order of their absolute impact values. Also, any additionally notable results are pointed out and to be discussed in Chapter 5. As described in Frischknecht (2020), all numerical LCIA results are rounded to three significant digits.

The results of each analysis stage is presented in a dedicated subchapter. To ensure comprehensiveness, each chapter restates the utilized FU and system boundaries of the respective analysis stage.

Above the uncertainty of electricity and heat generation as well as transport impacts has been identified. To facilitate the later discussion of the impacts of these flows, they are displayed as separate processes.

4.1 Attributional Analysis

As laid out above, the first analysis stage compares the impacts of all pathways for the first FU: 1 tonne of biowaste treatment. As an attributional analysis, the pathway's LCIs exclude any avoided processes from substituted product systems.

The pathways' impacts in the five impact categories are presented in five respective sub-figures. All sub-figures share the legend of Figure 6a. The impacts are broken down across the pathways' processes.

The impacts of the conventional pathways of food waste and grass clippings are substantially lower than the other pathways in most of the focus impact categories. Therefore, to improve readability, these pathways are excluded from the following figure. Furthermore, the impacts of the biogas pathways are found to be largely similar. Therefore, the co-digestion pathways are not depicted here but are only represented in the appendix in Figure A1.

Consequently, Figure 3 compares these three pathways: Conv.MM, Mono-AD and Grass-Gasf. for each impact category. According to the methodology of the hotspot analysis, all identified impact hotspots are mentioned. Also, a short conclusion is provided.

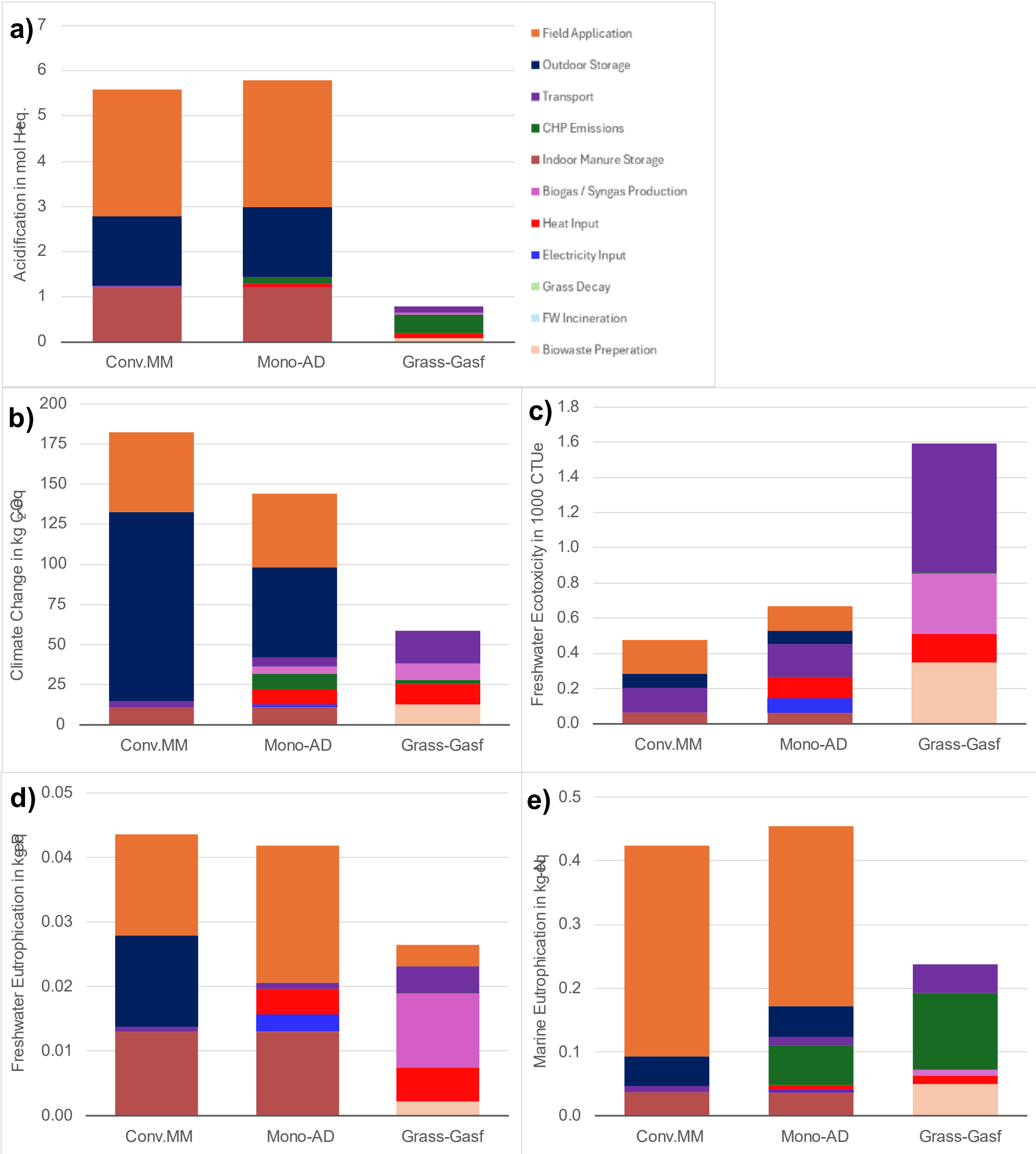

Figure 6: **Attributional impacts of key biowaste treatment pathways per tonne of biowaste.**
Source: Own data and design.

a) Acidification

The analysis reveals that the Grass-Gasf pathway causes by far the lowest acidification impacts. The Mono-AD pathway creates slightly higher acidification impact than the Conv.MM pathway. However, the impact profile of the Conv.MM and the Mono-AD pathways are very similar across processes. For both pathways, the most-relevant processes are their respective field application, digestate storage, and manure indoor storage processes. The additional processes from the production life cycle stage of the Mono-AD pathway make the

For both pathways, the ammonia emissions to the air are solely responsible for more than 80% of the acidification impacts. It is therefore the only elementary flow that is considered most-relevant by the hotspot analysis methodology by the European Commission (2021). The main occurrence of this significant flow is during manure and digestate storage, respectively.

In the Grass-Gasf. pathway the impacts from nitrogen oxide and sulphur dioxide are most relevant. These occur during the syngas combustion process, which is the single biggest contributor.

b) Climate Change

The Conv.MM pathway is assessed to cause the highest climate change impacts. The most-relevant elementary processes of the Conv MM pathway are the manure outdoor storage and field application stage. Methane and dinitrogen monoxide emissions are the most-relevant GHG-emissions.

Similarly, in the Mono-AD pathway, the digestate outdoor storage and field application processes are also found to be most-relevant alongside manure indoor management.

The methane emissions of the biogas pathway are less than half then in the conventional pathway. Dinitrogen monoxide has the single biggest contribution, followed by methane emissions, which are also most relevant.

In the Grass-Gasf pathway, the carbon dioxide is the only most-relevant elementary flow, which is emitted in various processes. Therefore, transport, heat input, biowaste preparation, and syngas production are the most-relevant processes.

c) Freshwater Ecotoxicity

The Grass-Gasf pathway is assessed to cause the most impacts in this impact category. Transport impacts, biowaste preparation, and syngas production are the most-relevant processes. Six elementary flows are most-relevant for these impacts: Aluminium III, iron ion, strontium II, hydrogen sulphide, anthracene, and nickel II.

The freshwater ecotoxicity impacts of the Conv.MM and Mono-AD pathways are both split among various processes. For the Conv.MM pathway, ammonia as well as aluminium III and Iron ions are most-relevant. The biogas pathway generates a little more freshwater ecotoxicity burden, due to heat and electricity inputs. Here the same elementary flows are most-relevant in addition to strontium II, and Hydrogen sulphide.

d) Freshwater Eutrophication

Once again, the Grass-Gasf pathway causes the least environmental impacts out of the three pathways. Syngas production, heat input, transport, and biochar field application are the most relevant processes. For this impact category, all impacts stem from phosphate emissions.

The Conv.MM pathway is assessed to cause slightly larger burden in freshwater eutrophication, than the biogas pathway. The field application, Outdoor storage and In-house storage of manure are the most relevant processes.

The biogas pathway is most impacted by the field application, and Indoor manure management processes. Notably, no impacts from the outdoor manure are identified.

e) Marine Eutrophication

For marine eutrophication, the Conv.MM and Mono-AD pathways are assessed to cause similar impacts, with field application and outdoor storage of the respective biowaste being the most-relevant processes. For the biogas pathway, the direct emissions during the CHP process is also a most-relevant process. Nitrate and ammonia are still the most-relevant elementary flows in both pathways, while nitrogen oxides are also most-relevant in the biogas pathway.

For the Grass-Gasf pathway, the CHP impacts, indoor manure management, and transport impacts are most-relevant. Nitrogen oxides is the only most-relevant elementary flow.

Conclusion

In conclusion, the Grass-Gasf pathway is the superior pathway in four of the five focus impact categories, while the Conv.MM and Mono-AD pathways cause similar impacts in most impact categories. The process impacts of the Conv.MM pathway are very similar to their equivalent process impacts in the Mono-AD pathway. The only apparent exceptions is the outdoor storage process in the acidification and climate change impact category. The additional processes of the Use stage of the biogas pathway are the tipping the scales in three out of five impact categories.

4.2 Consequential Analysis

This analysis compares the valorisation pathways. As in the previous analysis, the pathways are compared for FU1: treatment of 1 tonne of biowaste. However, in this analysis, each pathway includes its respective avoided biowaste treatments and outputs.

As in the analysis stage above, all identified impact hotspots are pointed out. The impacts in the five impact categories are presented in five respective sub-figures, which share a common legend presented in Figure 4a. Additionally, the sum of each pathway's impact is included in this analysis stage.

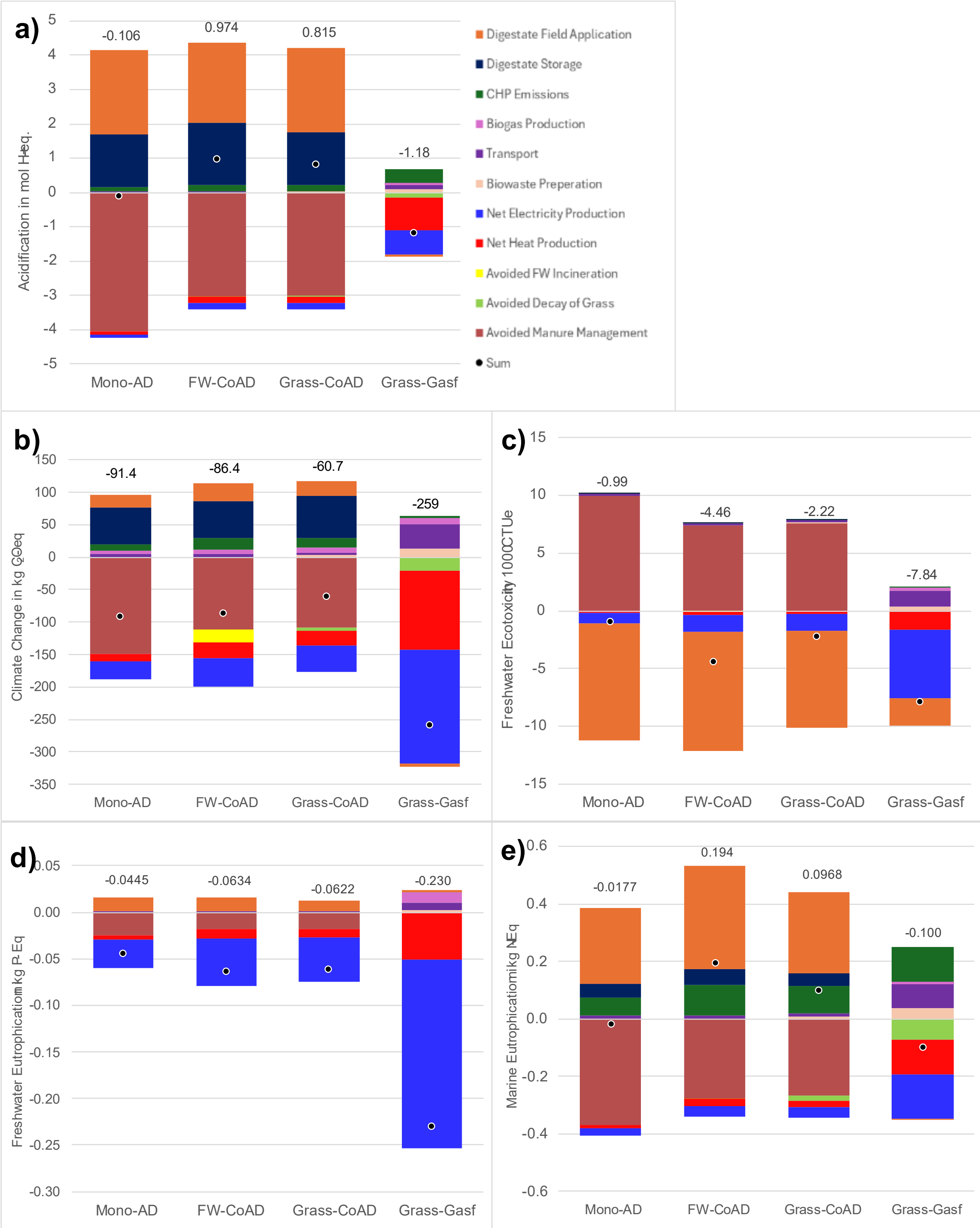

Figure 7: Consequential impacts of valorisation pathways per tonne of biowaste feedstock.
Source: Own data and design.

a) Acidification

The Mono-AD pathway is found to be the only biogas pathway to generate a slight environmental credit, while the co-digestion pathways result in a slight acidification burden.

For all three biogas pathways, the positive and negative acidification impacts are roughly balanced. The two co-digestion pathways cause a slight increase of impacts compared to the Mono-AD pathway and are therefore assessed to even cause more impacts then the conventional biowaste treatment pathways.

The biggest contributing processes of all three pathways are the avoided conventional manure management, while digestate field application and storage are also most-relevant. In this consequential analysis stage, the nitrogen oxide and sulphur dioxide emissions are the most-relevant elementary flows.

In the Grass-Gasf pathway, the impacts of the avoided products exceed the environmental burdens of the pathway. Besides net heat, biowaste preparation and CHP plant are the most-relevant processes. The most-relevant individual flows are the nitrogen oxide and sulphur dioxide emissions.

b) Climate Change

All biogas pathways generate environmental credit in the climate change impact category. However, the Mono-AD pathway ranks best among the biogas pathways. The inclusion of co-substrates in the biogas feedstock worsens the environmental footprint of the pathways. The food waste pathway produces a considerable amount of credit from avoiding conventional food waste incineration treatment, while the avoided grass decay plays only a minor role. The Grass-CoAD pathway again ranks as the least beneficial biogas pathway.

The processes with the biggest absolute contribution to the climate change impacts of all three pathways are the avoided manure management processes. For all biogas pathways, the impacts from the digestate storage and net electricity generation are also most-relevant. In the co-digestion pathways, the net heat generation is also most-relevant. The digestate field application is only most-relevant for the FW-CoAD pathway. For three pathways carbon dioxide and methane are the most-relevant elementary flows.

For the Grass-Gasf pathway, the only most-relevant processes are the net electricity and net heat generation. Here, only the carbon dioxide is most-relevant.

c) Freshwater Ecotoxicity

Just as in the climate change impact category, the implementation of all biogas pathway is assessed to result in an environmental benefit. This is especially prominent in the co-digestion pathways.

The processes that contribute most are the avoided manure management and the digestate field application. While the avoided manure management processes result in considerable environmental burdens, the digestate field application is the single biggest beneficial contributing process.

Ultimately, the FW-CoAD pathway is the most beneficial, while manure Mono-AD ranks as the least beneficial pathway.

In all biogas pathway, five elementary flows are most-relevant: hydrogen sulphide, aluminium (III), strontium II, chloride, and iron ion. Between the three biogas pathways, the ranking of these elementary flows only differs slightly.

Among the here assessed pathways the Grass-Gasf pathway is the most beneficial one. The net electricity is found to be the largest contributing processes, while biochar field application and net heat generation are also among the most-relevant processes. Five elementary flows are identified to be most-relevant: chloride, hydrogen sulphide, strontium II, aluminium (III) and iron ion.

d) Freshwater Eutrophication

All three biogas pathways result in a reduction in freshwater eutrophication impacts compared to their conventional biowaste treatments. The FW-CoAD pathway is assessed to be the most beneficial biogas pathway, while the Grass-CoAD pathway yields a similar amount of environmental credit. For each biogas pathway, the produced electricity has the single biggest influence on the impacts. Again, the field application and avoided manure management stages play a considerable role and are also most-relevant. As highlighted in the attributional analysis stage, phosphate is found to be the only elementary flow that is most-relevant.

For this impact category, the Grass-Gasf pathway is again assessed to be the most beneficial pathway. The environmental credits from the avoided electricity dominate the impacts from the other life cycle stages. This process and the net heat generation they are found to be most-relevant.

e) Marine Eutrophication

Across the life cycle stages, the marine eutrophication impacts are largely balanced between positive and negative contributions. The only biogas pathway that produces a slight environmental credit is the Mono-AD pathway. The Mono-AD pathway is found to possess two most-relevant processes: avoided manure management and digestate field application. The most-relevant elementary flows are nitrate and nitrogen oxides.

For the co-digestion pathways the CHP emissions are also fulfil the relevancy criteria. Also, ammonia is found to be most-relevant.

Conclusion

In conclusion, the Grass-Gasf pathway is the most beneficial pathway in all five focus impact categories. Among the biogas pathways, the Mono-AD pathway is assessed to be the most beneficial one in three out of five impact categories, while the FW-CoAD pathway is the most beneficial one in the remaining other two impact categories.

4.3 Context-based Analysis

Here, the results of the four municipal development scenarios are compared. This analysis stage is performed for FU2. Therefore, each development scenario includes the collective municipal biomass treatment of all three types of biowaste. It is assumed that a medium-sized biogas plant with an annual capacity of 20,000 tonnes of feedstock is operational and is supplied exclusively by either one of the three biogas pathways. The Grass-Gasf pathway is not included in any development scenario, as it is not mutually exclusive to any other valorisation pathways and deemed too immature for a timely implementation.

As identified in the hotspot analyses above, the impacts of the marine eutrophication are dominated by phosphate emissions. Despite the category's name, these are asserted to also be a regional environmental burden. Therefore, the results for this analysis stage are also considered here. The municipal carrying capacities are presented in the following Table 3.

Table 3: Carrying capacities of five impact categories in two sharing principles.

Focus Impact Category	Unit	Carrying Capacity "Equal per capita"	Carrying Capacity per "Land area"
Acidification	mol H+-Eq	1.32E+06	3.29E+06
Climate Change	kg CO_2-Eq	8.97E+06	2.24E+07
Freshwater Ecotoxicity	CTUe	1.73E+08	4.31E+08
Freshwater Eutrophication	kg P-Eq	7.65E+03	1.91E+04
Marine Eutrophication	kg N-Eq	2.65E+05	6.61E+05

Source: Own data and design.

These carrying capacities serve as benchmarks of absolute sustainability. They include the impact limits of all economic activities in the Hämeenkyrö municipality.

LCIA Results

The impacts of the Conv.Scenario primarily result from the conventional manure management. In four out of the five focus impact categories Conv MM causes over 90% of the respective impacts. Only in the climate change impact category do the FW-Incin and Grass-Decay pathways contribute more than 30 % of the impacts.

The following Figure 8 presents the impacts of the municipal development scenarios. The impacts are normalized by the carrying capacities based on the municipalities' share of the global population.

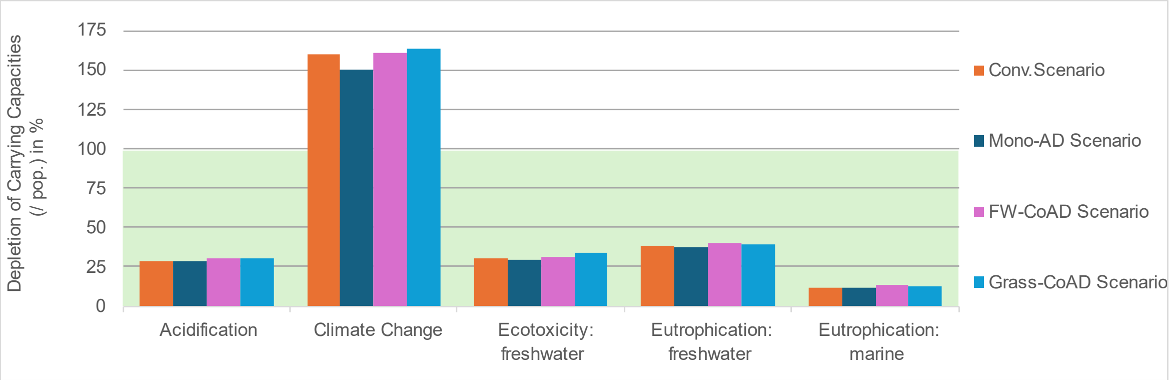

Figure 8: Impacts of development scenarios relative to the carrying capacities per municipal inhabitant. Carrying capacities are highlighted with a green background.
Source: Own data and design.

The analysis reveals that the four development scenarios produce similar levels of impacts in all five impact categories. No scenario clearly dominates the other scenarios.

While the Conv.Scenario produces the least acidification and marine eutrophication impacts, the Mono-AD Scenario is the most favourable scenario in the other three impact categories.

Three impact categories are strained for around one third of the carrying capacities. Notably the climate change impact category is clearly exceeded in all four scenarios. Conv.MM is responsible for around 87.1 % of these impacts. The implementation of the Mono-AD scenario is assessed to significantly decrease these climate change impacts. The marine eutrophication impacts are least depleted in all scenarios.

The following Figure 9 demonstrates the impacts of the development scenarios normalized by the carrying capacities of the second sharing principal "Land Area".

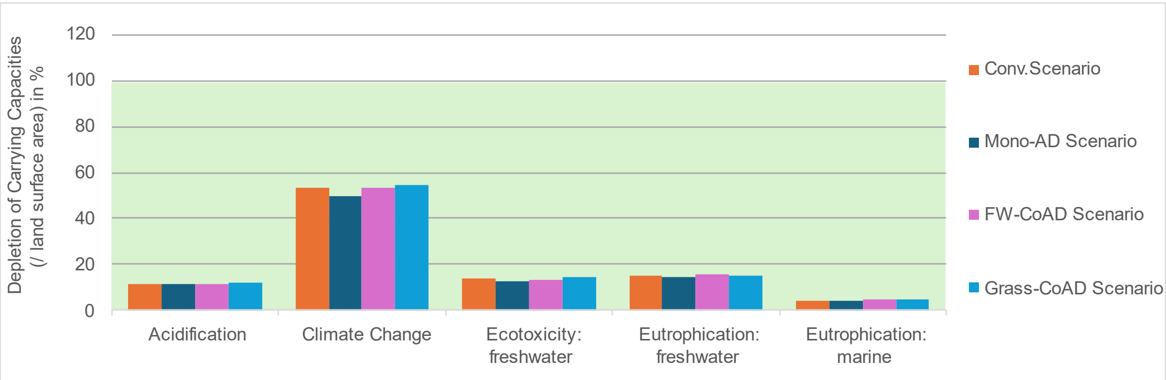

Figure 9: Impacts of development scenarios relative to carrying capacities per municipal land area. Carrying capacities are highlighted with a green background.
Source: Own data and design.

This analysis shows that this sharing principle considerably reduces the assessed relative depletion of the pathways. While the development scenarios rank in the same order as in Figure 8, the impacts of any scenario only deplete around half of the carrying capacities. As a result of that, all scenarios remain within the municipal impact limits.

The implementation of the Mono-AD Scenario reduces the municipality's overshoot of the carrying capacities. The carrying capacities in the PM, and climate change are assessed to be overshot. However, in sum with the other municipal economic activities, any impact category may still be exceeded.

4.4 Prospective Analysis

Lastly, the results of the pLCA are presented. It is found that the time frame does not play a significant role in the impacts of the Conv.MM pathway. Here, only minor changes in impacts from the transport process are noticeable.

First, an pLCA analysis is performed for FU1: treatment of 1 tonne of biowaste. The same system boundaries as in the consequential analysis in Chapter 4.2 apply. The results are shown in Figure 10.

To improve readability, the present-time impacts are only compared to the future impacts of time frame until 2050. This time frame is selected as it is the longest available time frame and also covers the total lifespan of a biogas plant that is assumed to be at least 20 years (Battini *et al.* 2014). A full overview of all four available time frames is provided in Figure A2.

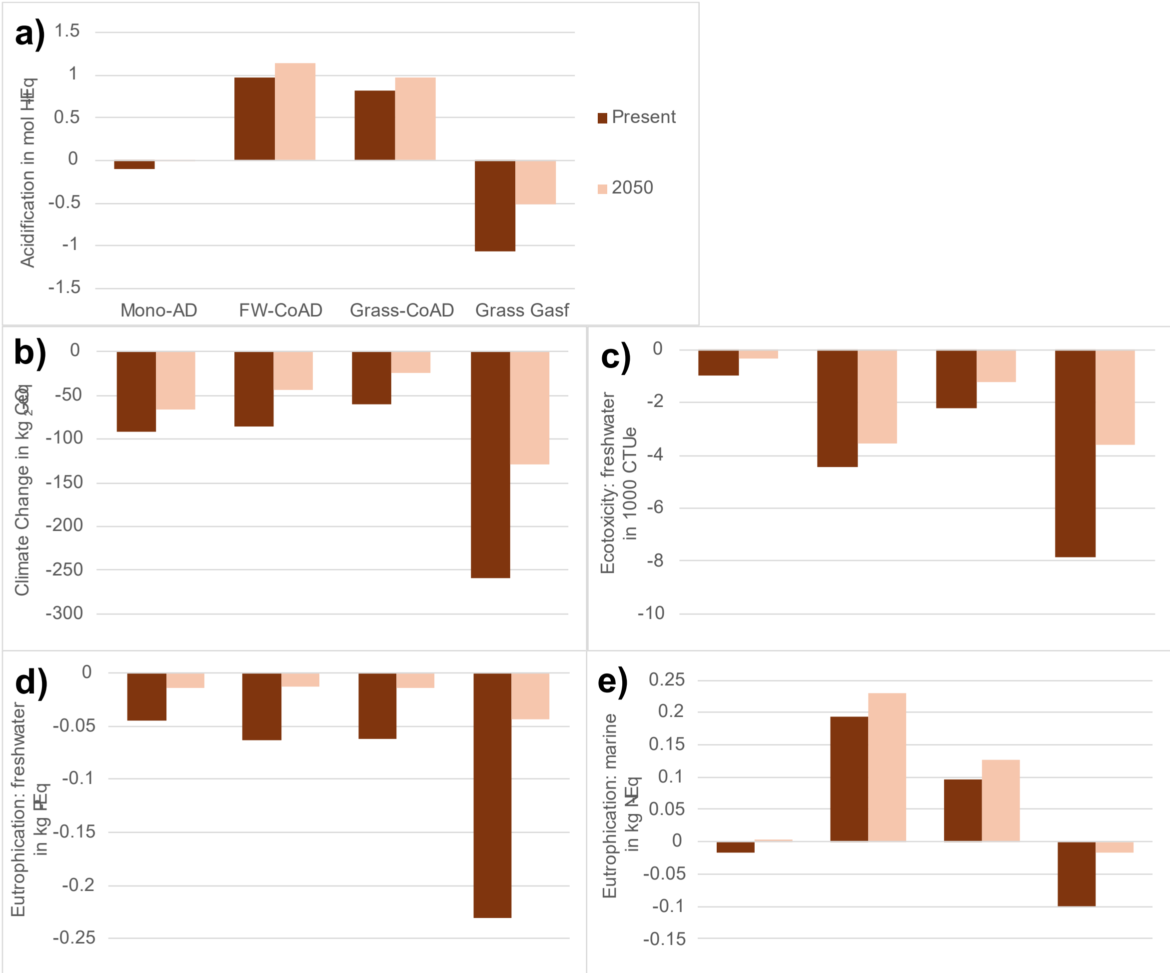

Figure 10: Consequential impacts of valorisation pathways per tonne of biowaste feedstock in two time frames.
Source: Own data and design.

In all analysed cases, the increased time frame affects the different impacts negatively. For example, the climate change credits of net electricity generation in the Mono-AD pathway decrease to less than 10% then in the recent time frame and less than 3% in the case of freshwater eutrophication. This analysis stage results in only one sign change for all impacts: Marine eutrophication of the Mono-AD pathway.

This general negative trend is especially prominent for the freshwater eutrophication impact category, where the created environmental credits reduce to around one fifth of the impacts assessed for the present time frame. This is also the only impact category, where this analysis reveals a change in the pathways' rankings By changing the time frame of the assessment, the FW-CoAD pathway becomes slightly inferior to the Grass-CoAD pathway, despite being slightly superior the consequential analysis in Chapter 4.2.

The switch to the 2050 time frame has the biggest absolute effect on the Grass-Gasf pathway. This is due to the assumed increase of regenerative energy in the Finnish electricity mix. Therefore, avoiding electricity production generates fewer environmental credits. Despite this considerable decrease in environmental credits, the Grass-Gasf pathway continues to be environmentally preferable to the biogas pathways in all impact categories.

The second prospective analysis shows the impacts for FU2: impacts of municipal biowaste treatment. The impacts of the municipal development scenarios of Chapter 4.3 are compared to the prospective impacts assessed up to 2050. As in Chapter 4.3, the assumption is made that a biogas plant with a feedstock treatment capacity of 20,000 tonnes per year is put into operation. Again, the analysis excludes any environmental credit of avoided conventional treatment pathways as well as of substituted products.

This pLCA of the context-based analysis stage finds insignificantly smaller than the impact of all scenarios compared to the present time frame. Therefore, these results are only depicted in the appendix in Figure A3 and Figure A4.

5 Discussion

In this chapter, the results of the SAF analysis stages are reviewed and interpreted. A complete evaluation by standard of a standardised LCA as described by the European Commission (2010) is beyond the limitations of this thesis. Instead, only the evaluation steps mentioned in Chapter 2.3.1.4 can be conducted.

First, the formalized hypotheses are compared to the SAF results. Secondly, key results of the LCI and LCIA are compared to literature values. Thirdly, the methodological limitations of the SAF are discussed and the suitability of the SAF methodology for this case study is assessed. Fourthly, practical hindrance of the EM and the valorisation pathway implementation are discussed and policy implications drawn.

5.1 Reflections on Hypotheses

This chapter summarises which of the hypotheses have been confirmed. Unconfirmed hypotheses are further discussed in the following chapters.

i. The SAF results confirm the first formulated hypothesis. All assessed biogas pathways are found to improve the environmental footprint of the Con. MM pathway in all five focus impact categories. Additionally, the Grass-Gasf pathway is found to decrease the environmental burdens of conventional grass decay.

ii. The consequential analysis stage assesses the co-digestion pathways to be inferior to the Mono-AD pathway in three of the five focus impact categories, while being superior in only two. Therefore, the second hypothesis is not confirmed.

iii. The context-based analysis stage partially confirms the hypothesis that the municipal impacts are reduced by employing biogas scenarios. However, this only applies to the Mono-AD scenario, as the results of the co-digestion scenarios are ambiguous and even increase the municipality's impacts in certain impact categories.

iv. The impacts from transportation are found to be irrelevant for most mostly pathways and impact categories. Therefore, the assumed transport reduction due to the EM is found to be largely insignificant, which contradicts the hypothesis.

v. The last hypothesis is partially confirmed, as by 2050 all pathways and biogas scenarios are assessed to produce higher environmental burdens or less environmental credits, respectively. Impacts of different biogas scenarios are also found to be slightly smaller.

5.2 Plausibility Checks

This chapter puts the calculated SAF results in their scientific context. First, the plausibility of the LCI is tested. After that, the results of this thesis' case study are compared with similar case studies. Subsequently, similar case studies are defined as having a similar object of interest, and goals and scope definition.

5.2.1 LCI Plausibility

As mentioned in Chapter 2.3.1.2, a feasible test of the LCIs plausibility is to verify the adherence to the laws of conservation of mass and energy. For that purpose, the adopted LCI data sources are revisited.

After that, the plausibility is further determined by comparing the calculated LCI data to the value range found in similar case studies. For that purpose, an exploratory literature review is conducted. The reviewed case studies are LCAs of cattle and pig manure mono- & co-digestion pathways at farm size scale in temperate climate.

5.2.1.1 Conservation of Mass and Energy

The adopted biogas LCI data from Hamelin *et al.* (2014) and Pehme *et al.* (2017) include mass balances and the energy values of all three biowaste types. Therefore, it can be asserted that the three biogas pathways adhere to the laws of conservation of mass and energy.

Nguyen and Hermansen (2015) also provide a comprehensive mass balance for the Grass-Gasf pathway. In addition, Voća *et al.* (2021) determined that the energetic value of grass is upwards of 16 MJ / kg. This is in line with the maximum potential electricity of 15,474 MJ / t calculated by Nguyen and Hermansen (2015) which adheres to the law of conservation of energy. Therefore, the laws of conservation of mass and energy are also confirmed for the Grass-Gasf pathway.

5.2.1.2 Key Inventory Results

In this chapter, the key inventory results presented in Table 2 are compared with the LCI results of similar LCA case studies. It should be noted that the LCIs of similar LCA studies subject(s) can still vary considerably, depending on individual modelling decisions and data accuracy (Esteves *et al.* 2019). It is outside the limitations of this thesis to determine whether the LCI values of this thesis are more realistic than the values found in the reviewed literature.

For the Grass-Gasf pathway, no inventory results are provided by Nguyen and Hermansen (2015). As no alternative comprehensive LCI data source is found in the literature, the plausibility of this pathway's results cannot be tested.

Feedstock

A review of manure co-digestion LCAs reveals that the share of food waste and grass in the feedstocks is in the range of the reviewed case studies. Out of all reviewed case studies, the relative share is found to be between 0.25 and 0.363 (El-Mashad and Zhang 2010; Zhang *et al.* 2013).

Anaerobic Digestion

The Mono-AD pathway is modelled to yield 35 Nm³ of biogas. For comparison purposes, literature biogas yields are adjusted to represent a methane content of 65 %. Özdemir (2025) found the mono-digestion of cattle manure to produce around 26.9 m³ biogas. Zhang *et al.* (2015) found that a similar process in Canada yielded around 33.5 Nm³.

The LCI of this thesis reveals that the utilization of co-substrates increases the biogas yield of the anaerobic digestion by at least 50 %. This is in line with Esteves *et al.* (2019), who find an increase of biogas yield of 36 % to 96 % across different co-digestion LCAs.

CHP

The electricity and heat generation efficiencies at the CHP plant are less commonly reported in the reviewed case studies. From the CHP characteristics presented in Battini *et al.* (2014) a net electricity generation of around 172 MJ per tonne of cattle manure can be calculated. Eggemann *et al.* (2023) found a similar electricity generation efficiency.

This study finds that only 5 % to 12 % of generated electricity is consumed during the biogas pathway. This is in line with the used LCI data sources Hamelin *et al.* (2014) and Pehme *et al.* (2017). However, in a review of different biogas LCA studies, Huttunen *et al.* (2014) found that the biogas pathway requires between 20 % and 50 % of the biogas energy content. This is a significant difference to the LCI results of this case study. Further technical research should be undertaken to determine the origins of this difference and to verify the assumptions made in the cited LCA case studies.

Also, LCI data on net heat generation is scarcely provided in the reviewed literature. Some case studies, like Battini *et al.* (2014), even assume that no heat can be exported from the biogas pathway. The LCI data of Eggemann *et al.* (2023) can be calculated to generate a net heat of 5.58 MJ / Nm³ of biogas. This latter result is in the range of the net heat efficiencies calculated for this case study, which range from 4.43 MJ / Nm³ to 5.95 MJ / Nm³ of biogas.

Construction

The LCI of the SAF does not include the construction or disposal of farm infrastructure, as they are tested to have a lower impact than 1 % on the LCIA results for each of the FUs. This decision is therefore in line with the technical system boundary mentioned in Chapter 2.3.1.1.

Summary

The key inventory results appear to mostly be in line with similar case studies. Only, the share of self-consumed energy is a major point of contention.

5.2.2 LCIA Plausibility

This chapter seeks to verify the plausibility of the SAF results. A focus lies on the total impacts of the pathways and the identified most-relevant processes and elementary flows. The plausibility of the attributional results is not tested explicitly, as they are part of the consequential results and are least relevant for policy decision making.

First, the plausibility of the SAF results of the consequential pathways is tested. After that, the results of the context-based analysis stage are put in scientific context, followed by a plausibility test of the prospective results.

5.2.2.1 Consequential Pathway Results

The plausibility of the pathway result is tested. This is especially important for the total pathway results, as they are relevant to the further analysis stages. This plausibility test also serves to identify possible mistakes during the practical modelling steps that should be fixed in future iterations of this case study.

Finding comparable LCIA data is proven to be more difficult than finding comparable LCI results. This is due to the additional layer of differentiation from the goal and scope definition, and the methodological decisions. Therefore, LCIA results of biogas studies with similar LCIS can still vary widely. Possibly for this reason, literature reviews of manure-based biogas LCA studies such as Esteves *et al.* (2019) or Bacenetti *et al.* (2016) do not include numerical comparisons of LCIA results.

The only other source of comparable pathway LCIA results is found in the original LCI data source studies. Consequently, the pathway results are compared to LCIA results of the LCI data source studies outlined in Table 1.

As pointed out above, no LCI data source study exists for the FW-CoAD pathway. However, the results of the FW-CoAD pathway can be seen as plausible as its results are in range of other biogas pathways.

Range of GHG Impacts

Pehme *et al.* (2017) assessed their pathways to cause similar impact profiles across processes for most impact categories. However, for the global warming impact category, they find the range of total pathway impacts to be more than twice as wide as calculated in this consequential analysis stage. As Pehme *et al.* (2017) does not provide the impacts of individual elementary flows, these differences cannot be traced to the elementary flows level.

The identified differences in total pathway impacts cannot plausibly be ascribed to the slight difference in the study subject(s) or the goal and scope definition. While it is outside the limits of this thesis to assert the cause of this discrepancies, two methodological decisions that may cause this difference are examined followingly.

First, this could be due to the different LCIA methods employed: While this thesis employs the EF method, Pehme *et al.* (2017) used a combination of the EDIP2003 and IPCC 2013 methods (Hauschild and Potting 2005; IPCC 2013). Therefore, different classification processes are conducted and different CFs used, among others.

Another possible cause is the different background LCI database used. This case study employs the premise database, while all three LCI data source studies utilize some version of the Ecoinvent LCI database (Weidema *et al.* 2013). The premise database is less detailed and contains fewer processes than the Ecoinvent database. The smaller number of processes likely decreases the size of the assessed product system model and their impacts.

Further reflections about the premise database are presented in Chapter 5.3.3.

Process Impacts

The literature reviews of biogas LCA case studies find that some processes commonly cause impact hotspots (Bacenetti *et al.* 2016; Esteves *et al.* 2019). To further test the plausibility of the LCIA results, it is checked whether the calculated LCIA results have similar impact hotspots across the assessed processes.

Transportation

Esteves *et al.* (2019) point to the high variability of the impacts from transportation in different LCA case studies of manure-based biogas. Bacenetti *et al.* (2016) found that transports are one of the main impact contributors when biowaste is transported over long distances.

This contrasts with consequential LCIA results, as transport emissions are found to be nearly irrelevant. A sensitivity analysis should be conducted for different transportation distances. However, this is beyond the limitations of this thesis.

AD Plant

Esteves *et al.* (2019) highlight the high relevance of the AD plant processes to phosphate emissions, resulting in a share of up to 25 % of acidification and eutrophication impacts. Contrary to that, Bacenetti *et al.* (2016) point out that impacts from the AD plant only play a minor role in the total pathway impacts.

This case study assessed that the AD plant causes an insignificant impact share in almost all impact categories and pathways.

Digestate Storage and Field Application

Esteves *et al.* (2019) find that anaerobic digestion of manure is beneficial due to reducing emissions during storage and field application, among others. Bacenetti *et al.* (2016) also find that methane leaks during storage and unharnessed methane are significant to the climate change impact category. These finding are largely confirmed by this case study.

Conclusively, the case study results are in line with the findings of the literature reviews. It is found to be notable, how few common impact sources have been identified between the case studies reviewed by Esteves *et al.* (2019) and Bacenetti *et al.* (2016). This reinforces the above-mentioned notion regarding the high variability of biogas LCIA results based on the study design.

Bioenergy Impact Comparison

In this chapter, the results of the consequential analysis stage are compared to the LCIA results of the LCI data source studies. As mentioned above, the LCI data sources do not provide attributional results of their conventional pathways. Therefore, the plausibility of the conventional pathways cannot be tested. However, some insights into the plausibility of the Conv.MM results are provided in Chapter 5.2.2.2. Also, as part of the following subchapters, the share of the avoided conventional pathways in the valorisation pathway is discussed.

Biogas Pathways

Pehme *et al.* (2017) does not allow for a comparison of LCIA impacts in the acidification impact category, as a different impact indicator is used. Furthermore, Pehme *et al.* (2017) does not calculate ecotoxicity impacts.

A comparison of the consequential results and the results of Pehme *et al.* (2017) show a similar distribution of impacts across processes for the Mono-AD and Grass-CoAD pathways.

However, a notable difference is identified in the relation of total climate change impacts of the grass co-digestion pathway compared to the manure mono-digestion: In this thesis' case study, the Grass-CoAD pathway generates significantly less environmental credits than the Mono-AD pathway. This is not in line with the literature cited above and Pehme *et al.* (2017), which assessed this pathway to generate significantly more environmental credits relative to the Mono-AD. This difference can be traced to different assessments of the grass decay process.

Throughout all biogas pathways and impact categories, the share of digestate storage impacts from the total pathway impacts are assessed to be higher than in Pehme *et al.* (2017). As mentioned above, the origin of this discrepancy cannot be traced to the elementary flow level, as no flow data is provided by the compared case studies.

Based on this LCIA comparison, the identified process impacts cannot be deemed plausible and should be verified in future iterations of this thesis.

Grass Gasification

Comparison of this case studies result to the results of its LCI data source requires adapting the FU and system boundaries of Nguyen and Hermansen (2015). The comparison of grass gasification pathway impacts reveals that the LCI source study assessed vastly different climate change impacts than this case study. While this case study determined that the Grass-Gasf pathway generates environmental credits, Nguyen and Hermansen (2015) found that environmental burdens are produced. Nguyen and Hermansen (2015) also claims that three other case studies all assess climate change burden.

in this case study, the climate change credits from heat production are much more relevant than in Nguyen and Hermansen (2015). Also, this study finds that the impacts from biowaste collection and preparation are way less significant than in the LCIA data source study.

The plausibility of the consequential results of the climate change impact category cannot be asserted. This might be due to e.g. the difference in used background database, or the used LCIA method.

The three other impact categories that are also part of this case studies focus impact categories are assessed to be negative. However, they cannot be compared numerically, as they are calculated in a different impact indicator.

5.2.2.2 Context-based Results

The context-based analysis stage utilized a novel methodological approach, which is specific to the analysed municipality. Therefore, the calculated carrying capacities cannot be compared numerically to literature values. The employed sharing principles are discussed as part of Chapter 5.3.3.

The context-based analysis showed that conventional cattle manure management alone exceeds municipal carrying capacity of climate change of all economic sectors of the municipality. As this is an unexpected result, a small series of rough plausibility calculations are conducted to identify the potential error source.

Conv.Scenario Impacts

First, the plausibility of the provided municipal MFA data is tested. Based on Kallio *et al.* (2025), the most prominent cattle breed in Finland is Holstein. Weiss and St-Pierre (2010) calculated that a herd of a hundred Holstein cows produces a daily amount of around 8,940 tonnes of manure. Finish national statistics assert that Hämeenkyrö keeps around 3,984 head of cattle (LUKE 2023). Therefore, the annual cattle manure production of cattle manure is around 65,300 tonnes. This is roughly in line with the biowaste data provided in the MFA, which reported an annual amount of to 71,040 tonnes (SDU 2024). As mentioned above, the MFA is reported to also account for some biowaste from other municipalities. As this result is plausible, the MFA is eliminated as a responsible error source.

Secondly, it is examined to what extent the impacts of the Conv.MM pathway are responsible for the unexpectedly high Conv.Scenario impacts. For that purpose, the climate change impacts from conventional manure management in Hämeenkyrö are compared to the national impacts from manure management. This relation can then be compared to the relation of municipal and national cattle livestock.

The Finish manure management annually emits around 700 kt of CO_2-equivalents (Statistics Finland *et al.* 2024) and is dominated by raw manure application (Bywater *et al.* 2025).

The MFA assessed that Hämeenkyrö produces 71,040 tonnes of manure per year (SDU 2024), which equates to an annual total of 12.4 kt of CO_2-equivalents. These impacts correspond to around 1.84 % of the national manure management emissions of all livestock types.

The total number of Finish cattle is around 764,400 (LUKE 2025). Therefore, the municipal share on the total Finish cattle livestock is around 0.52 %.

Consequently, the municipal cattle herds cause more than three times the GHG impacts compared to their share of Finnish manure management emissions. This is implausible, especially regarding the fact that Hämeenkyrö also possess significant number of other livestock (LUKE 2025).

Assuming that the above presented municipal and national statistics about emission and livestock data are correct, this only leaves the results of the Conv.MM impacts to be potentially flawed. This plausibility check suggests that the impacts of Conv.MM are at least three times above expected levels, which makes them implausible. Notably, the plausibility of the Conv.MM impacts could not be verified in Chapter 5.2.2.1.

If it is therefore assumed that the impacts of the Conv.MM pathway are only a third of the calculated results, this would have grave implications for the results of the conventional scenarios. Under this assumption, the impacts from the Conv.Scenario would deplete the carrying capacities (per capita) only to around 50 %. Consequently, further research is required to verify the impacts of Conv.MM.

The impacts of the Conv.MM are relevant for all pathways in the consequential analysis stage, as well as for all development scenarios in the context-based analysis stage. Future research should investigate whether the results of the Conv.MM are sufficiently regionalized to the Finnish context.

5.2.2.3 Prospective Results

To test the plausibility of the prospective results, the assumed share of regenerative energy in the Finish national grid mix in the premise database is investigated. It is revealed that the modelled Finnish electricity mix is not a regionalised process but only a proxy of the European electricity grid mix. Consequently, the three single biggest electricity sources in the Finish market are modelled to be French nuclear energy, German coal energy and German onshore wind energy.

By comparison, the Ecoinvent database contains a regionalized market process of the electricity mix in Finland. In this market process, the single biggest energy sources besides electricity imports into Finland are nuclear, hydro, and wind energy, which have low environmental impacts (Weidema *et al.* 2013).

A comparison of one MJ of avoided electricity between the Ecoinvent and premise database reveals that, depending on the focus impact category, only 4 to 40 % of the environmental credit are generated. An overview of these impacts is provided in Figure A6.

Therefore, during all analysis stages, the environmental credits generated from avoiding conventional electricity generation are exaggerated. As part of that, in the prospective analysis stage, the difference of environmental impacts between the present and 2050 time frame are actually lower than assessed. This makes the results of the prospective analysis stage appear implausible.

This finding is curious because, as mentioned-above, the premise database is a modified version of the Ecoinvent database (Bruhn *et al.* 2023). Therefore, the electricity processes have been assumed to be equivalent.

This difference is not due to missing characterisation, as the “LCIA Check” function in the openLCA software contains similar inventory results.

This finding is especially peculiar, as electricity is of especial relevance to this pLCA approach. As mentioned above, in the premise database, the different time frames are modelled by altering the market mixes of energy products. Future research should implement more regionalized electricity market mix processes in premise.

Next, the plausibility of prospective SAF results is further testes. For that purpose, a comparison with other prospective LCIA results is attempted. In the scope of this thesis, no similar pLCA study on the investigated bioenergy pathways could be identified. However, Pehme *et al.* (2017) conducted a sensitivity analysis of the mono-digestion pathway, which includes an "avoided wind energy" scenario.

In this scenario it is assumed that the avoided electricity is exclusively generated through wind energy. This electricity generation can be assumed to closely resembles the electricity mix assumed for the 2050 time frame in the premise database.

In this "avoided wind energy" scenario significantly less climate change credits are generated from the avoided net electricity. This effect is most prominent in the impact category corresponding to freshwater eutrophication, where nearly all credits disappear. This effect is also discovered in this pLCA in the 2050 time frame.

5.3 Model and Data Limitations

Various limitations of this study's results regarding the utilized data and methodologies can be pointed out. Those should be discussed to facilitate this case study's replication for other municipalities.

In this chapter, sources of uncertainty are identified and tested base on literature suggestions. Sensitivity analyses are conducted for some identified sources of uncertainty. Due to the context-based approach of the SAF, a focus lies on the defined geographical, temporal, and technological system boundaries. The findings are structured along the first three LCA phases, as well as a separate discussion on the novel LCA approaches.

5.3.1 Goal and Scope

It is uncertain whether the modelled product systems are (still) relevant to real life municipal context of Hämeenkyrö. This is especially questionable, as the employed product model and LCI data originate from LCA case studies published between 8 - 11 years before this thesis.

Following, some identified sources of product model inaccuracy are pointed out. However, in the scope of this thesis, only one sensitivity analysis can be conducted, which is presented last.

Firstly, MTK (2025b) argue that composting is already a relevant pathway of food waste treatment in Finland. In the scope of this thesis, the current relevance of this pathway cannot be asserted. Secondly, Finnish climate conditions are found to lead to additional practical problems of bioenergy systems. For example, during winter, various biowaste types can be frozen (Huttunen *et al.* 2014), which would require an additional defrosting process in the product system.

Thirdly, regional stakeholders assume that newly constructed biogas storage facilities will feature a closed roof instead of a straw cover (MTK 2025b). Closed storage has been found to considerably reduce the direct storage emission (Battini *et al.* 2014; Bacenetti *et al.* 2016).

Further iteration of this case study should assess the importance of these differences and adapt the product system accordingly.

Uncertain of Energy Outputs

Stakeholder interviews conducted by Huttunen *et al.* (2014) revealed high uncertainty of the context-depended amounts of energy. Stakeholder reported that feeding electricity into the grid is generally not profitable. Also, in some cases, the processing of biowaste in the biogas plants is reported to consume "all the electricity produced" (Huttunen *et al.* 2014: p.11).

On a similar note, Bacenetti *et al.* (2016) found that in some unspecified cases manure-based biogas plant are found to not have significant heat outputs. This is reported to be a relevant case in Finland, as heat demand is low during summer (Huttunen *et al.* 2014).

The attributional and consequential analysis stages of the SAF reveal that these assumptions influence the environmental impacts of the biogas pathways. In Figure 6 it is assessed that even if no energy outputs are generated, the biogas pathways cause no considerable environmental damages (under the assumption that manure and digestate cause equivalent impacts). However, this assumption changes the rankings between the three biogas pathways.

Sensitivity Analysis of Biomethane Upgrading

It is commonly discussed that, from an economic and environmental perspective, biogas should not be combusted in a CHP plant but instead be upgraded to biomethane (Huttunen *et al.* 2014). By increasing the methane content of the biogas, biomethane is produced that can be fed into the natural gas grid. In that way, a fossil fuel is directly substituted (Ravi *et al.* 2023).

This is especially relevant as the prospective analysis stages has revealed that the environmental credits of electricity avoidance decrease over time. Meanwhile, natural gas is likely to still be relevant in 2050, and therefore also its substitution by biomethane. Therefore, the following sensitivity analysis is conducted:

For that purpose, the environmental impacts of 1 m^3 of biogas utilization in two processes are compared in a consequential gate-to-gate scenario analysis. It is assumed that a purification plant is built on farm-site with access to the natural gas network.

The CHP process is modelled according to Pehme *et al.* (2017), which has been also used in this thesis' case study. The biomethane upgrading process is the most popular upgrading technology in Europe according to the premise database: "biomethane, high pressure {CH}| biogas purification to biomethane by amino washing | Cut-off, U".

The amount of net electricity generation itself has above been identified as a relevant source of sensitivity. Therefore, this sensitivity analysis includes an additional CHP scenario in which no net electricity is generated. To examine potential prospective changes, each scenario is also run for the 2050 time frame. The following figure presents the result of the sensitivity analysis for the five focus impact categories.

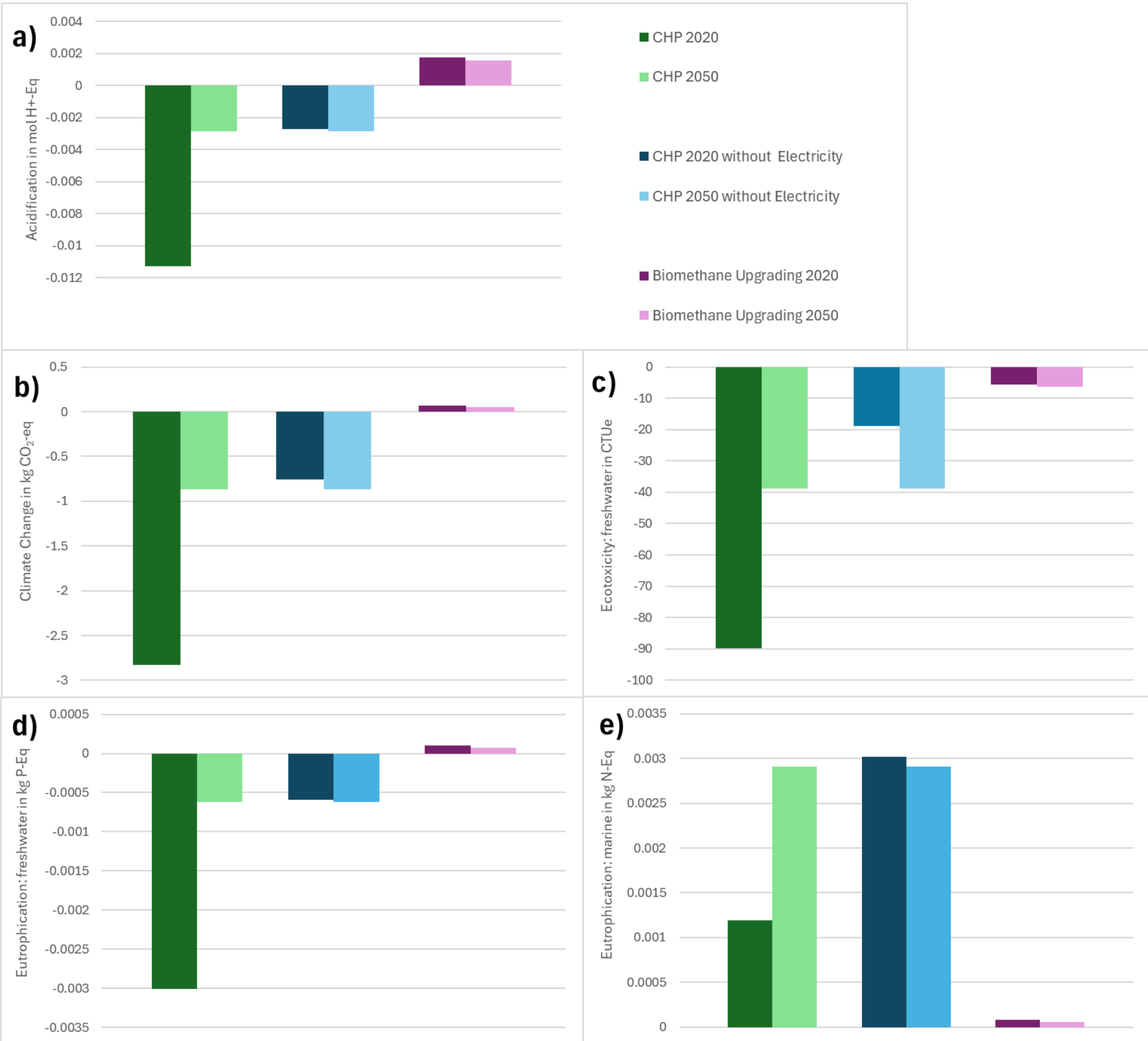

Figure 11: Sensitivity analysis for biogas utilisation scenarios in two time frames.
Source: Own data and design.

This sensitivity analysis reveals that the biomethane upgrading scenario causes the most impacts in four of the five focus impact categories. Further exploration of the impacts of the biomethane upgrading process reveals that this process consumes a significant amount of heat input. The impacts from heat input play a significant role in all impact categories but freshwater eutrophication.

This finding is surprising, as it contradicts the assertions made above. However, this finding is in line with other LCA case studies like Ravi *et al.* (2023), who also found that biomethane upgrading is not advantageous to CHP in any of the assessed impact categories, except for "resource use, fossil". They conclude that the share of regenerative energy in the national electricity mix plays a significant role in that comparison, which is also found in the case conducted above.

This sensitivity analysis revealed that the biomethane upgrading process generates fewer environmental credits than the CHP scenario. This is unintuitive and cannot be explained logically. It can be assumed that this finding might be due to the immaturity of the premise database. Future research should verify the amount of required heat input and rerun this sensitivity analysis in another background database.

In the 2050 time frame, the higher impacts of the biomethane upgrading process shrink. Just as above, this finding can be explained by the assumed increase of renewable energy in the national electricity and heat mix.

5.3.2 Life Cycle Inventory

In this chapter, some prevalent limitations of this study's results regarding its LCI are presented and their relevance to the LCIA results is evaluated. The limitations discussed relate to the data quality of the transferred LCI, the sensitivity of the LCIA results to the employed allocation rule, and finally regarding the employed EoL rule.

Reliance on Generic Data

As mentioned above, the LCI methodology discourages the transfer of previously compiled LCI data into a new LCA case study, as potential error sources are also transferred. This is especially relevant for this thesis' case study, due to the potential outdatedness of the LCI data source studies. Following, some identified issues of the transferred LCI data are identified.

First, all LCI data source studies rely on modelled to calculate LCI data instead of experimental data. However, real life LCI data may differ from modelled LCI data. For example, Hossain *et al.* (2023) compared modelled and experimental LCI data for a biogas LCAs case and found impact differences between the two methods of up to 100 %. This is especially relevant as ex-ante LCAs of biogas plants necessarily rely on modelled LCI data.

Secondly, the transferred LCI data is based on biochemical characteristics of the biowaste. These characteristics are potentially not representative for the Finnish case. This is especially relevant for the cattle manure, as food waste and grass cuttings are assumed to be comparable across countries. However, this concern is limited by the fact that the same cattle breed of Holstein cows is dominant in Estland and Finland (Kallio *et al.* 2025). Therefore, the manure composition is argued to be similar.

Thirdly, it is found that in the LCI data source studies the biogas yields are calculated based on the partial biogas potential of the different biowaste types. Critically, this methodology does not account for the different C / N ratio in the different feedstocks. This is problematic because, as mentioned above, it is found that an optimal C / N ratio increases the biogas yield. This potentially compromises the comparison of biogas pathways as improving the C / N ratio is one of the main motivations behind co-digestion. This error source may partially be responsible for the unexpectedly weak performance of the co-digestion pathways.

Sensitivity to Allocation Rule

In accordance with the relevant ISO norms, the product systems of this thesis exclusively employ the system expansion approach (DIN 2006). However, as stated in Chapter 2.3.1.4, a sensitivity analysis becomes mandatory when an allocation has been conducted.

Therefore, a sensitivity analysis is conducted in which the economic allocation approach is employed instead of the system expansion approach. In the economic allocation approach, substituted products are not modelled as avoided products. Instead, the economic value of the products is assessed, and subsequent allocation factors are calculated.

Based on the European Commission and Eurostat (2025) the average annual retail price of high-voltage electricity in Finland in 2024 was 78.8 € / MWh. As of 2023, the average price of district heat was found to be 91.2 € / MWh (Ranta *et al.* 2025). Due to the limitations of this thesis, all biowaste is assumed to have no economic value and therefore also no allocation factors.

A dedicated sensitivity analysis for the consequential analysis stage and the context-based analysis stage is performed. Additionally, a short conclusion is provided alongside some remarks on the economic allocation approach.

Sensitivity of the Valorisation Pathways

The following figure illustrates the sensitivity of the consequential analysis stage to adopting an economic allocation approach instead of a system expansion approach. The same system boundaries as in Chapter 4.2 apply. All pathways are calculated for FU1: 1 tonne of biogas feedstock.

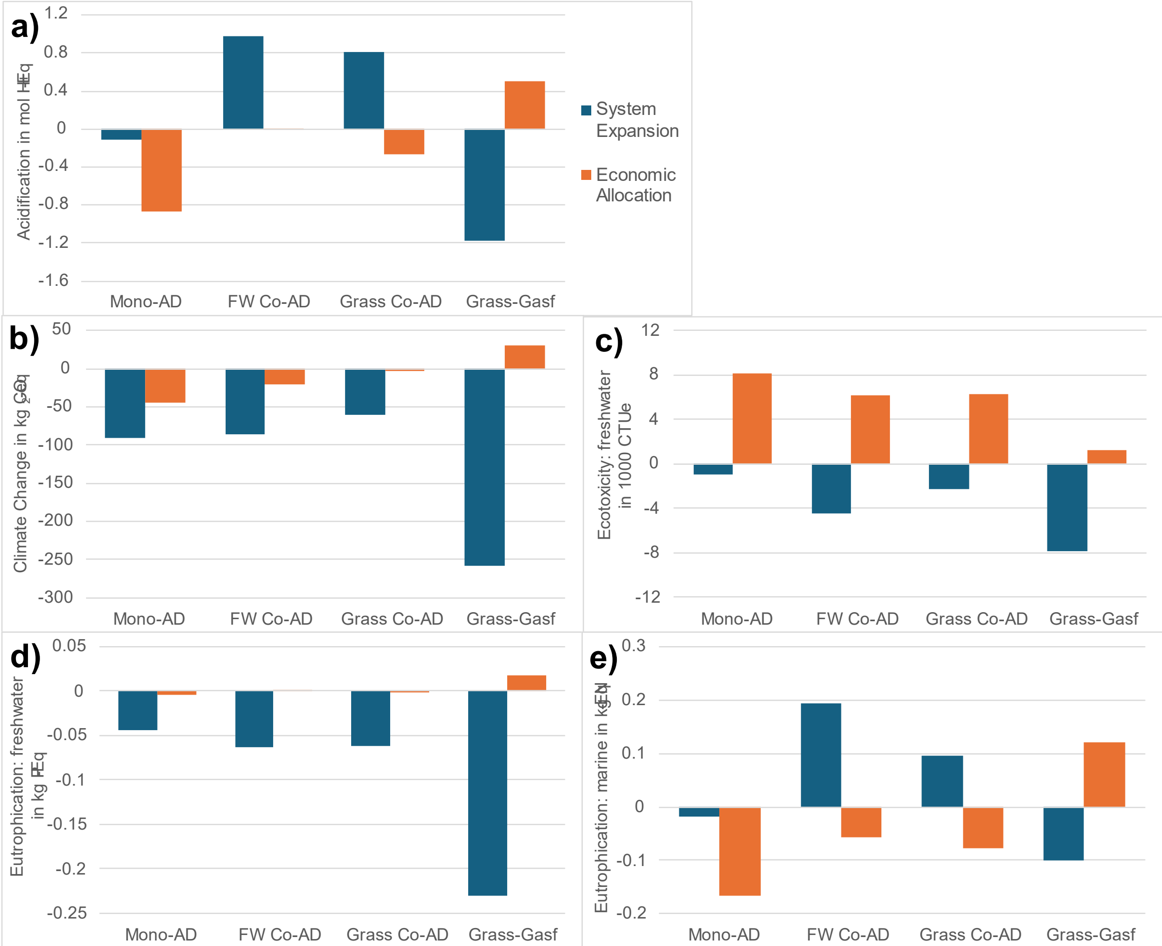

Figure 12: Impacts of the valorisation pathways for two different allocation rules in the five focus impact categories.
Source: Own data and design.

This sensitivity analysis reveals that switching to an economic allocation approach has major implications for almost all impacts. While the impacts of some impacts are reduced by up to a factor of four, other impacts are vastly increased. The biggest changes are found in the Grass-Gasf pathway. This is curious, as this pathway has the biggest share of avoided products in its total pathway impacts. However, in the scope of this thesis, the cause of these discrepancies cannot be identified.

Sensitivity of the Municipal Development Scenarios

A second sensitivity analysis is performed for the three biogas scenarios in the context-based analysis. As in the previous analysis, the system expansion approach is exchanged for an economic allocation approach. The same system boundaries and the same FU as in Chapter 4.3 apply.

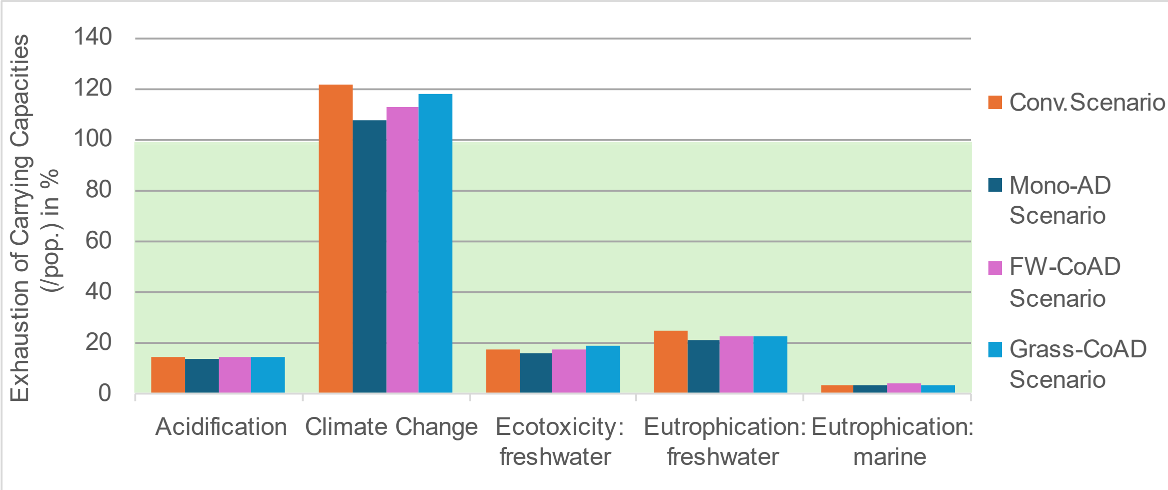

Figure 13: impacts of municipal development scenarios as percentages of the carrying capacities per municipal inhabitant. Employing economic allocation.
Carrying capacities are highlighted with a green background.
Source: Own data and design.

When employing an economic allocation approach, the depletion of the carrying capacities is assessed to be lower than assessed for the system expansion approach in Figure 8. The biggest change is in the climate change depletion which is reduced by approximately a fourth. These results are mostly due to a reduction of impacts in the conventional pathways. Additionally, the differences between scenarios become slightly more pronounced. The assessed depletion of the freshwater eutrophication carrying capacity is least affected by this methodological change.

Finally, the sensitivity of the context-based analysis with the land area sharing capacities is tested.

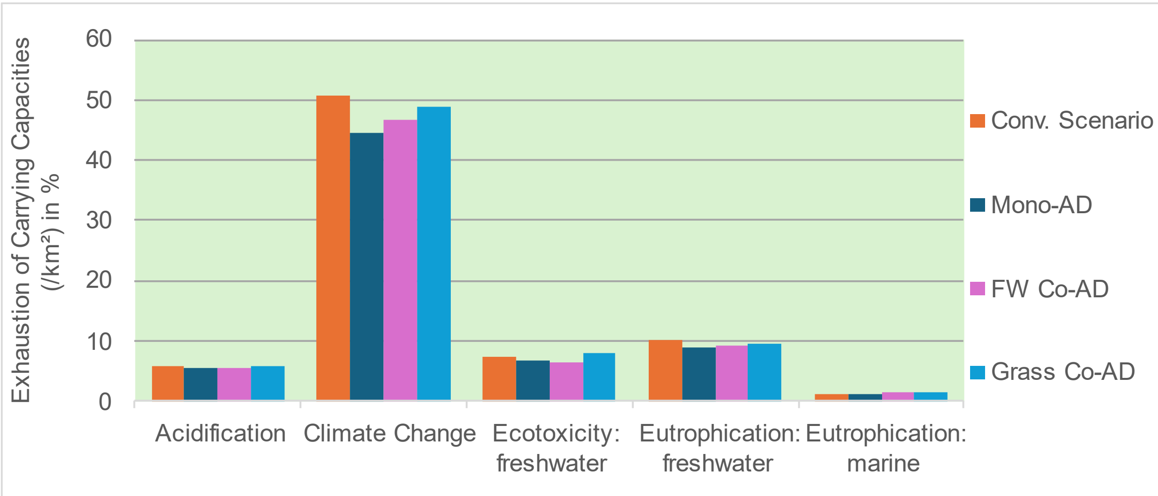

Figure 14: impacts of municipal development scenarios as percentages of the carrying capacities per municipal land area. Employing economic allocation.
Carrying capacities are highlighted with a green background.
Source: Own data and design.

The results reveal that each scenario depletes only around 50 % of the climate change carrying capacity. The other carrying capacities are mostly depleted by less than a tenth, which is the lowest depletion out of all assessed context-based analyses.

This constellation of the context-based analysis is notable, as it found the lowest depletion ratios of all assessed context-based analyses of this case study. Throughout all analyses, the depletion of the climate change carrying capacity ranges from 160 % (system expansion, "equal-per-capita") and 50 % (economic allocation, "land area").

Conclusion

This sensitivity analysis revealed a high sensitivity of all pathways and development scenarios to employing the economic allocation approach. This is a relevant finding , as this allocation approach is highly relevant for LCA practice (Jungbluth 2023).

However, it should also be noted that the economic allocation approach also has some theoretical disadvantages. For example, future price changes also affect the allocation factors, which adds an additional layer of uncertainty to the results (Ardente and Cellura 2012). This is especially relevant for prospective impacts, as prices are likely to change over the longer time periods.

End-of-Life Rule

The employed premise database uses a Cut-off approach to model the EoL of recycled products. As detailed Chapter 2.3.1.2, no impacts of a material's first life cycle is attributed to its second life cycle. However, some case studies use databases which employ other EoL rules like the APOS approach. In this approach a share of a materials first life cycle is allocated to the second life cycle of the product (Ekvall *et al.* 2020).

The different EoL rules are especially relevant for this case study as it involves open-loop-recycling and would therefore cause changes to the result. However, currently only a Cut-Off premise database is available. Future research should conduct a sensitivity analysis to assess the importance of the EoL rule.

5.3.3 Life Cycle Impact Assessment

This chapter discusses the relevance of impacts from impact categories, besides the five focus impact categories. For that purpose, the results of the attributional Conv.MM and Mono-AD pathway are revisited.

The EF methods features a so-called single score. In the single score normalisation factors are used to make the results of different impact categories comparable. However, the normalisation factors also add another layer of uncertainty to the LCIA results. Therefore, the SAF does not include single score results in its analysis stages.

Still, the rankings of the normalisation results of the EF method reveal that, besides the five focus impact categories, other impact categories are also highly affected. The following subchapters explore the relevance of those.

Particulate Matter Formation

This impact category assesses the negative human health effects caused by particulates smaller than 2.5 μm (Andreasi Bassi *et al.* 2023).

In the Conv.MM and Mono-AD pathways, the particulate matter formation impacts are among the most affected impact categories. Its impacts stem almost exclusively from ammonia emissions, which occur during the digestate field application and storage processes. This result is unexpected, as particulate matter formation is not emphasised as a relevant impact category in any reviewed literature reviews (Huttunen *et al.* 2014; Bacenetti *et al.* 2016; Esteves *et al.* 2019). Further research is required to verify this result.

Land-Use

In the EF LCIA method, the Land-Use impact category indicates impacts to soil quality, which is assessed via biophysical soil parameters (Andreasi Bassi *et al.* 2023). It should therefore not be confused with the concepts of land use change or indirect land use change.

However, the foreground model of this case study's LCI does not include the effect of manure and digestate fertilization on soil quality. These are assumed to have high land use impacts, which can be assumed to far exceed the calculated land-use impacts. Therefore, it is asserted that this impact category is unsuitable for this case study. Future research should include these soil quality changes to generate more meaningful land-use results.

The normalised land-use impacts of the consequential analysis are not striking compared to other impact categories. This is in accordance, with the few cases in which this impact category in included (Bacenetti *et al.* 2016; Esteves *et al.* 2019).

5.3.4 Novel LCA approaches

This case study tested two novel LCA approaches. Here, their utility and suitability for this case study is reflected upon.

5.3.4.1 Context-based Analysis

In this chapter, findings of additional impact categories are discussed. After that, the accuracy of the calculated carrying capacities is reflected upon. Lastly, a reflection on the implementation of PB in this case study is made.

Other Impact Categories

Outside the five focus impact categories, the carrying capacities of particulate matter and land-use are found to be the most depleted. The Conv.Scenario depletes these carrying capacities by around 300 % and 50 %, respectively. An overview of the context-based analysis stage results including these two impact categories is provided in Figure A5.

Further research should seek to identify the extent of particulate matter formation impacts and their relevance to the carrying capacities. However, for this case study, the land-use impact category is asserted to only be of limited meaning, as detailed above.

"Fair" Carrying Capacities

A normative value judgment is made as soon as a set of carrying capacities and a sharing principle is selected (Bjørn *et al.* 2020). In this chapter, other calculation approaches as well as other sustainability benchmarks are examined.

The utilized sharing principle "Equal per capita" is found to be the most commonly used sharing principle among AESA studies reviewed by Bjørn *et al.* (2020). The Land area approach is found to be very rarely employed, potentially due to little applicability for non-regional case studies (Bjørn *et al.* 2020).

An example of another applicable sharing principle is the "Historical debt/ability to pay" principle. In this principle, the global carrying capacities are shared based on historical emissions. Therefore, European regions like Hämeenkyrö would likely be assigned very small shares (Heide and Gjerris 2024). As this sharing principle would decrease the carrying capacity of Hämeenkyrö, its relative depletions would consequently increase.

Further, the carrying capacities provided by Sala *et al.* (2020) are a potential source of uncertainty. For example, Bjørn and Hauschild (2015) also developed an alternative set of normalisation factors based on different global emission limits. Further research could employ these to test the sensitivity of the regional carrying capacity depletion.

PB in LCA

The PB concept might not be fully appropriate for a regional case study, as some impact categories like freshwater eutrophication have predominantly regional impacts. Impacts within the calculated regional limits might overlook regional impact vulnerability (Paulillo and Sanyé-Mengual 2024).

However, Paulillo and Sanyé-Mengual (2024) identified an increasing interest in implementing PB in LCA, especially in AESA. However, they also found relevant mismatches between the control variables of the PB framework and the indicator units of frequently used comprehensive LCIA methods. Therefore, further synchronization is required (Paulillo and Sanyé-Mengual 2024).

An alternative option of implementing PB in LCA is to employ PB impact categories. For example Ryberg *et al.* (2018) compiled a dedicated PB-LCIA method that can be used instead of conventional LCIA methods.

This approach avoids the above-mentioned mismatch of PB control variables and LCIA indicator units. However, this less tested PB-LCIA method creates a new source of uncertainty (Ryberg *et al.* 2018). To employ a PB-LCIA method in this case study, compatibility with the premise database would first need to be established.

5.3.4.2 Prospective LCA Analysis

This chapter discusses the potential of improving the pLCA analysis by introducing scaling factors. Above, some unexpected results are ascribed to the utilized premise database. This chapter provides a potential explanation for that.

Scaling Factors

Most of the above-mentioned limitations of the SAF compound in the prospective analysis. For example, the relevance of the used data and the assessed product system is likely to decrease in the 2050 time frame due to technical innovations.

This problem could be partially mitigated by introducing scaling factors to the foreground LCI. These can be used to include technological efficiency improvements to key inventory flows. Due to the limitations of this thesis, no suitable scaling factors could be implemented.

Ironically, due to the agedness of the LCI data sources, the inclusion of scaling factors might have already been sensible for the present time frame. Further research should employ adequate scaling factors in the foreground model.

Database Implementation

Bruhn *et al.* (2023) point out that the data format of the premise modification tool is incompatible with the openLCA software. However, a first implementation of the premise database to openLCA has recently been accomplished. Therefore, this thesis is assumed to be one of the first case studies to utilize the premise database in openLCA.

As part of this first implementation, potential technical errors might have occurred. These might have caused a faulty LCI database, which would explain some of the unexpected results identified above regarding the background database. Future research should seek to implement a premise database to the open software to identify possible mistakes of this thesis' implementation.

5.4 Unlocking EM Potential

This case study highlights the environmental potential of EM for biowaste. However, this potential largely depends on farmers, who function as buyers and sellers of bioenergy feedstock and side streams.

However, Ngo (2023) and MTK (2025c) identified a timid uptake of the current EM implementation. This is unsurprising as timid uptake is a common phenomenon in their initial phases for an EM (Taghipour *et al.* 2021). This is also in line with experiences from other CE transitions, as new CE pathways are known to requires socio-institutional change in "written and unwritten rules, customs and beliefs" (Potting *et al.* 2017: p.6).

Some concrete hindrances with attracting new EM users have been identified and presented in a workshop conducted by MTK (2025c). These are particularly relevant to this case study, as they could limit the implementation of the assessed bioenergy pathways and scenarios.

Furthermore, increasing the EM uptake would also unlock additional bioenergy potential, as the here assessed biogas scenarios only contain less than a third of the municipal cattle manure. Additional biowaste types, like pig manure, are also a commonly discussed bioenergy feedstock (Esteves *et al.* 2019). To unlock the bioenergy potential of the municipality, policymakers should aim to adapt to the practical barriers of EM. Two examples are presented below.

Antitrust Issues

Antitrust issues can emerge among market participants due to uncertainty about the products offered. Relevant uncertainties include for example, the bioenergy potential or the viscosity (MTK 2025c).

It has been documented that in an EM, the complexity of a product and its necessity to have a detailed product description makes it more likely for purchasers to limit their number of suppliers (Taghipour *et al.* 2021). This reduces the potential efficiency of the current biowaste EM. Therefore, software developers should allow for product reviews functions, which serve as a feedback mechanism for other market participants (Taghipour et al., 2021).

Digital Fatigue

Finnish farmers report that they show symptoms associated with digital fatigue (MTK 2025c). This condition, which “arises from prolonged engagement with digital tools” (Supriyadi *et al.* 2025: p.1). It describes a feeling of overload with digital tool in the workplace. A common symptom is the loss of productivity during working online (Supriyadi *et al.* 2025). It can therefore be assumed that affected farmers are less inclined to utilize the EM to the full extend.

Therefore, policymakers and software developer should encourage face-to-face interactions between market participants (Taghipour *et al.* 2021), for example in communication or during bio-waste quality assessment. A mandatory offline transaction procedure may be suitable.

Additionally, the EM should strive to create a convenient and time effective user experience. Therefore, the implementation of the mobile access may be especially relevant, as farmers predominantly use mobile devices for internet access (MTK 2025c). Therefore, a practical EM should feature a convenient mobile functionality and user interface.

6 Conclusions

The case study largely confirms the formulated hypotheses about the environmental potential of establishing a biowaste EM. It has been proven that environmental burdens from biowaste treatment in the Hämeenkyrö municipality can potentially be reduced.

It is assessed that the investigated bioenergy pathways generate various environmental credits by avoiding conventional biowaste treatment and fossil fuel usage. Anaerobic digestion is found to be an environmentally beneficial alternative to raw cattle manure application in all impact categories. The mono-digestion of cattle manure is identified as the most beneficial utilisation of biogas capacity regarding acidification, climate change, and marine eutrophication impacts. Co-digestion of manure with food waste proved to be the most beneficial pathway regarding freshwater ecotoxicity and freshwater eutrophication

For most assessed bioenergy pathways, biowaste storage and field application are common emission hotspots, which cause significant ammonia and nitrogen oxide emissions. The results of the grass gasification technology are found to be even more beneficial in all impact categories. However, uncertainty remains regarding a large-scale implementation.

The regional assessment of different biogas scenarios led to ambiguous results. Only utilizing manure in mono-digestion results in a reduction of environmental impacts in climate change, freshwater ecotoxicity, and freshwater eutrophication impact category. However, acidification, and marine eutrophication impacts increase. Employing a co-digestion scenario is found to even increase the environmental impacts in all these impact categories. The context-based LCA approach is found to be a valuable extension of the LCA methodology, while also being highly sensitive to methodological changes.

In the time frame until 2050 the environmental credits of all pathways are projected to decrease, due to the assumed increase in the share of regenerative energy in the regional electricity grid. The pLCA approach is found to be a useful sustainability assessment methodology. However, the employed implementation of the premise database appeared to be immature and proved to be a source of uncertainty.

Most pathway results are found to be in a plausible impact range. A notable exception is the conventional manure management pathway, which are potentially overestimated. This raises general concerns about the validity of biogas results that include the substitutions of this pathway.

Various data and methodological limitations have been identified throughout the SAF. Therefore, this case study cannot provide a meaningful basis for policymaking. Further research should seek to validate the used LCI data. Replications of this case study in other municipalities should also explore other highlighted methodological approaches.

Still policymakers should strive to realize the biowaste EM potential by mitigating the effects of digital fatigue and antitrust among farmers. To improve the circularity of biowaste, policymakers should be alert to further technological improvements that have higher environmental benefits like biomass gasification.

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Appendix

I. Foreground LCI Process Data

Following, the process LCI data of the foreground model of consequential analysis stage are presented. For completeness reasons, values are not rounded. First, the used parameters are shown.

Table A1: Parameters

Name	Value
CH4_C	1.335442511
CO2_C	3.664141204
eleceff_syngas	0.38
EMef	0.5
EMu	0.1
heateff_syngas	0.57
K2O_K	1.204529029
N2O_N	3.143785714
NH3_N	1.2165
NO3_N	4.428571429
NO_N	2.142857143
P2O5_P	2.29147571
PO4_P	3.066561188
scf_CHP_elec	0.4
scf_CHP_heat	0.46

Conventional Pathway Processes

Processes are sorted by pathways. Following, the processes of the conventional pathways are displayed. After that, the processes of the valorisation pathways are presented. All flow values represent the process data for the consequential analysis stage. A negative amount indicates a waste flow. Processes that are part of multiple pathways are only displayed once.The provider column indicates flow connections to other processes.

Conv.MM

Table A2: **Manure Indoor Storage.**

Inputs	Category	Waste Flow	Amount	Unit	Provider
manure FA		x	1000	kg	
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		0.36*NH3_N	kg	
Carbon dioxide, biogenic	Emissions to air/low. pop.		0.015*CO2_C	kg	
Dinitrogen monoxide	Emissions to air/low. pop.		0.0126*N2O_N	kg	
manure, liquid			1087	kg	
Methane, biogenic	Emissions to air/low. pop.		0.019*CH4_C	kg	
Nitrogen	Emissions to air/low. pop.		0.011	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.000165*NO_N	kg	
Phosphate	Emissions to water		0.0139*PO4_P	kg	

Table A3: **Manure outdoor storage**

Inputs	Category	Waste Flow	Amount	Unit	Provider
manure FA	_model	x	1000	kg	
manure, liquid	_model		1000	kg	Manure Indoor Storage
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		0.42*NH3_N	kg	
Carbon dioxide, biogenic	Emissions to air		2.11*CO2_C	kg	
Dinitrogen monoxide	Emissions to air/low. pop.		0.0232*N2O_N	kg	
manure FA	_model	x	1100.276	kg	Manure Field Application
Methane, biogenic	Emissions to air/low. pop.		2.73*CH4_C	kg	
Nitrogen	Emissions to air/low. pop.		0.009*2	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.000142 * NO_N	kg	
Phosphate	Emissions to water		0.0139*PO4_P	kg	

Table A4: Manure Field Application

Inputs	Category	Waste Flow	Amount	Unit	Provider
inorganic nitrogen fertiliser, as N	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-2.105*	*kg*	*2020_SSP2_NPi_inorganic nitrogen fertiliser, as N {RER}| nutrient supply from calcium ammonium nitrate | Cut-off, U*
inorganic phosphorus fertiliser, as P2O5	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*- (1.09 - 0.0139) * 0.6 * P2O5_P*	*kg*	*2020_SSP2_NPi_inorganic phosphorus fertiliser, as P2O5 {RER}| nutrient supply from diammonium phosphate | Cut-off, U*
inorganic potassium fertiliser, as K2O	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*- 3.68 * 0.9 * K2O_K*	*kg*	*2020_SSP2_NPi_inorganic potassium fertiliser, as K2O {RER}| nutrient supply from potassium chloride | Cut-off, U*
liquid manure spreading, by vacuum tanker	processing/Agricultural/Transformation		1	m3	2020_SSP2_NPi_liquid manure spreading, by vacuum tanker {CH}| liquid manure spreading, by vacuum tanker | Cut-off, U
manure FA	_model	x	1000	kg	
transport, tractor and trailer, agricultural	transport/Road/Transformation		10-10*0.5*Emu	t*km	2020_SSP2_NPi_transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		0.69*NH3_N	kg	
Carbon dioxide, biogenic	Emissions to air/low. pop.		40.9*CO2_C	kg	
Dinitrogen monoxide	Emissions to air/low. pop.		(0.042+0.0068+0.0016)*N2O_N	kg	
Nitrate	Emissions to water/groundwater		0.2154*NO3_N	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.0042 * NO_N	kg	
Phosphate	Emissions to water		0.0139*PO4_P	kg	

FW-Incin

Table A5: Food Waste Incineration.

Inputs	Category	Waste Flow	Amount	Unit	Provider
electricity	*energy/Electricity country mix/High voltage/Market*		*-74*	*MJ*	*2020_SSP2_NPi_electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U*
Food Waste	_model	x	1000	kg	
heat	*energy/Heat/Others/Market*		*-0.21*	*MJ*	*2020_SSP2_NPi_heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U*
transport	transport/Road/Market		15	t*km	2020_SSP2_NPi_transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U
Outputs	Category		Amount	Unit	Provider
Ammonia	Emissions to air/low. pop.		0.00000293	kg	
Ammonia	Emissions to soil/industrial		1.58E-10	kg	
Ammonium, ion	Emissions to water/river		0.0000329	kg	
Antimony, ion	Emissions to air/low. pop.		5.4E-10	kg	
Arsenic, ion	Emissions to air/low. pop.		0.000000577	kg	

Benzene	Emissions to air/low. pop.		7.02E-08	kg	
Cadmium (II)	Emissions to air/low. pop.		9.94E-08	kg	
Carbon dioxide, biogenic	Emissions to air		520	kg	
Carbon dioxide, fossil	Emissions to air/low. pop.		63.4	kg	
Carbon monoxide, biogenic	Emissions to air/low. pop.		0.0812	kg	
Chloride	Emissions to water/river		0.447	kg	
Chromium (III)	Emissions to air/low. pop.		0.00000212	kg	
Cobalt (II)	Emissions to air/low. pop.		4.48E-08	kg	
Copper, ion	Emissions to air/low. pop.		0.00000155	kg	
Dinitrogen monoxide	Emissions to air/low. pop.		0.000658	kg	
Dioxin, 2,3,7,8 Tetrachlorodibenzo-p-	Emissions to air/low. pop.		3.4E-10	kg	
Hydrocarbons, unspecified	Emissions to air/low. pop.		0.00466	kg	
Hydrogen chloride	Emissions to air/low. pop.		0.03	kg	
Hydrogen fluoride	Emissions to air/low. pop.		0.0000622	kg	
Hydrogen sulfide	Emissions to air/low. pop.		0.000000281	kg	
Lead II	Emissions to air/low. pop.		0.000000865	kg	
Manganese (II)	Emissions to air/low. pop.		0.000000294	kg	
Mercury (II)	Emissions to air/low. pop.		0.000000624	kg	
Methane, biogenic	Emissions to air/low. pop.		0.362	kg	
Nickel II	Emissions to air/low. pop.		0.00000361	kg	
Nitrogen oxides	Elementary flows/Emission to air/low population density		1.4	kg	
NMVOC, non-methane volatile organic com-pounds	Elementary flows/Emission to air/low population density		0.009	kg	
PAH, polycyclic aromatic hydrocarbons	Emissions to air/low. pop.		2.82E-08	kg	
Particulate Matter, > 2.5 um and < 10um	Emissions to air		0.00213	kg	
Selenium (IV)	Emissions to air/low. pop.		0.00000395	kg	
Sulfate	Emissions to water/river		0.0154	kg	
Sulfur dioxide	Elementary flows/Emission to air/low population density		0.075	kg	
Titanium, ion	Emissions to air/low. pop.		9.73E-11	kg	
Vanadium (V)	Emissions to air/low. pop.		0.00001	kg	
Zinc (II)	Emissions to air/low. pop.		0.00000244	kg	

Grass Decay

Table A6: Grass Decay.

Inputs	Category	Waste Flow	Amount	Unit	Provider
Grass, decaying	_model	x	1000	kg	
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Carbon dioxide, biogenic	Emissions to air		0.4812471	t	
Dinitrogen monoxide	Emissions to air/low. pop.		0.074130435	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.182518116	kg	

Valorisation Pathway Processes

Mono-AD

Table A7.: Anaerobic Digestion – Mono-AD.

Inputs	Category	Waste Flow	Amount	Unit	Provider
electricity	energy/Electricity country mix/High voltage/Market		4.5	kWh	2020_SSP2_ high voltage {FI}| market for electricity, high voltage | Cut-off, U
heat	energy/Heat/Others/Market		130	MJ	2020_SSP2_NPi_ district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U
manure, liquid			1000	kg	Manure Indoor Storage
transport	transport/Road/Transformation		0	t*km	
Outputs	**Category**		**Amount**	**Unit**	**Provider**
biogas	E:Water supply; sewerage		35	m3	
Carbon dioxide, biogenic	Emissions to air		0.357	kg	
Digestate, MonoAD		x	959.5	kg	Storage Mono-AD
manure FA		*x*	*-1000*	*kg*	*Manure Outdoor Storage*
Methane, biogenic	Emissions to air/low. pop.		0.163	kg	

Table A8: Digestate Storage Mono-AD.

Inputs	Category	Waste Flow	Amount	Unit	Provider
Digestate, MonoAD	_model	x	1000	kg	

electricity	energy/Electricity country mix/High voltage/Market		2.9	kWh	2020_SSP2_NPi_electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		0.435*NH3_N	kg	
Carbon dioxide, biogenic	Emissions to air		2.572	kg	
Digestate, MonoAD	_model	x	1111.4	kg	Field Application Digestate Mono-AD
Dinitrogen monoxide	Emissions to air/low. pop.		0.0335*N2O_N	kg	
Methane, biogenic	Emissions to air		1.206	kg	
Nitrogen	Emissions to air		0.0111	kg	
Nitrogen oxides	Emissions to air/low. pop.		1.73E-4*NO_N	kg	

Table A9: Digestate Field Application Mono-AD.

Inputs	Category	Waste Flow	Amount	Unit	Provider
Digestate, MonoAD	_model	x	1000	kg	
inorganic nitrogen fertiliser, as N	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-3.031*	*kg*	*2020_SSP2_NPi_inorganic nitrogen fertiliser, as N {RER}| nutrient supply from calcium ammonium nitrate | Cut-off, U*
inorganic phosphorus fertiliser, as P2O5	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*- 0.672 * P2O5_P*	*kg*	*2020_SSP2_NPi_inorganic phosphorus fertiliser, as P2O5 {RER}| nutrient supply from diammonium phosphate | Cut-off, U*
inorganic potassium fertiliser, as K2O	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-3.278 * K2O_K*	*kg*	*2020_SSP2_NPi_inorganic potassium fertiliser, as K2O {RER}| nutrient supply from potassium chloride | Cut-off, U*
liquid manure spreading	processing/Agricultural/Transformation		1	m3	2020_SSP2_NPi_liquid manure spreading, by vacuum tanker {CH}| liquid manure spreading, by vacuum tanker | Cut-off, U
transport	transport/Road/Transformation		10-10*Emu	t*km	2020_SSP2_NPi_transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Outputs	**Category**		**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		0.7136*NH3_N	kg	
Carbon dioxide, biogenic	Emissions to air		106.7	kg	
Dinitrogen monoxide	Emissions to air/low. pop.		0.05048*N2O_N	kg	
Nitrate	Emissions to water/groundwater		0.18252*NO3_N	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.00433 * NO_N	kg	
Phosphate	Emissions to water		0.0197*PO4_P	kg	

Table A10: CHP Biogas

Inputs	Category	Waste Flow	Amount	Unit	Provider
biogas	E:Water supply; sewerage		0.044*incl_P	m3	Parameter
electricity, high voltage	*energy/Electricity country mix/High voltage/Market*		*-0.4*	*MJ*	*2020_SSP2_NPi_electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U*
heat	*energy/Heat/Others/Market*		*-0.35522*	*MJ*	*2020_SSP2_NPi_heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U*
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Carbon dioxide, biogenic	Emissions to air		0.0836	kg	
Carbon monoxide, biogenic	Emissions to air/low. pop.		0.00031	kg	
Dinitrogen monoxide	Emissions to air/stratosphere + troposphere		0.0000016	kg	
electricity, high voltage	_model		0.4	MJ	
Methane, biogenic	Emissions to air/low. pop.		0.000434	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.000202	kg	
NMVOC, non-methane volatile organic compounds	Elementary flows/Emission to air/low population density		0.00001	kg	
Particulate Matter, < 2.5 um	Emissions to air/low. pop.		0.000000206	kg	
Particulate Matter, > 2.5 um and < 10um	Emissions to air/low. pop.		0.000000245	kg	

FW-CoAD

Table A11: Food Waste Hygienisation.

Inputs	Category	Waste Flow	Amount	Unit	Provider
Electricity	energy/Electricity country mix/High voltage/Market		10.6	kWh	2020_SSP2_NPi_ electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U
Transport	transport/Road/Transformation		15	t*km	transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U
Food Waste	*_model*	*x*	*-1000*	*kg*	*Food Waste Incineration*
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Food Waste	_model		960	kg	

Table A12: Anaerobic Digestion – FW-CoAD

Inputs	Category	Waste Flow	Amount	Unit	Provider
Electricity	energy/Electricity country mix/High voltage/Market		1.459560432	MJ	2020_SSP2_NPi_ electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U
Food Waste	_model		250.411	kg	Food Waste Hygenisation
Heat	energy/Heat/Others/Market		128.07	MJ	2020_SSP2_NPi_ heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U
manure, liquid	_model		749.588	kg	Manure Indoor Storage
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
biogas	E:Water supply; sewerage		59.24553073	m3	
Carbon dioxide, biogenic	Emissions to air		0.49700483	kg	
Digestate, FW AD	_model	x	931.393675	kg	Storage FW-CoAD
manure FA	*_model*	*x*	*-749.588*	*kg*	*Manure Outdoor Storage*
Methane, biogenic	Emissions to air/low. pop.		0.2761138	kg	

Table A13: Digestate Storage FW-CoAD

Inputs	Category	Waste Flow	Amount	Unit	Provider
Digestate, FW AD	_model	x	1000	kg	
Electricity	energy/Electricity country mix/High voltage/Market		2.9	kWh	2020_SSP2_electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U
Outputs	**Category**		**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		0.530199585*NH3_N	kg	
Carbon dioxide, biogenic	Emissions to air		3.351921198	kg	
Digestate, FW AD	_model	x	1018.2	kg	Field Application FW-CoAD
Dinitrogen monoxide	Emissions to air/low. pop.		0.020517034*N2O_N	kg	
Methane, biogenic	Emissions to air		1.635741813	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.000221016*NO_N	kg	

Table A14: Field Application Digestate FW-CoAD

Inputs	Category	Waste Flow	Amount	Unit	Provider
Digestate, FW AD	_model	x	1000	kg	
inorganic nitrogen fertiliser, as N	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-4.4*	*kg*	*2020_SSP2_NPi_inorganic nitrogen fertiliser, as N {RER}| nutrient supply from calcium ammonium nitrate | Cut-off, U*
inorganic phosphorus fertiliser, as P2O5	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-0.787553384 * p2O5_P*	*kg*	*2020_SSP2_NPi_inorganic phosphorus fertiliser, as P2O5 {RER}| nutrient supply from diammonium phosphate | Cut-off, U*

inorganic potassium fertiliser, as K2O	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-3.611025869 * k2O_K*	*kg*	*2020_SSP2_NPi_inorganic potassium fertiliser, as K2O {RER}| nutrient supply from potassium chloride | Cut-off, U*
liquid manure spreading	processing/Agricultural/Transformation		1	m3	2020_SSP2_NPi_liquid manure spreading, by vacuum tanker {CH}| liquid manure spreading, by vacuum tanker | Cut-off, U
transport	transport/Road/Transformation		10-10*Emu	t*km	2020_SSP2_NPi_transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		(0.024366882+0.754830804)*NH3_N	kg	
Carbon dioxide, biogenic	Emissions to air/low. pop.		30.31613511*CO2_C	kg	
Dinitrogen monoxide	Emissions to air/low. pop.		(0.062902567+0.007854879)*N2O_N	kg	
Nitrate	Emissions to water/groundwater		(0.002328256+0.310434075)*NO3_N	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.006290257* NO_N	kg	
Phosphate	Emissions to water		0.023101566*PO4_P	kg	

Grass-CoAD

Table A15: Grass Collection.

Inputs	Category	Waste Flow	Amount	Unit	Provider
diesel	energy/Mechanical/Market		142	MJ	2020_SSP2_NPi_diesel, burned in agricultural machinery {EUR}| diesel, burned in agricultural machinery | Cut-off, U
Transport	transport/Road/Transformation		9.5	t*km	transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Grass, decaying	*_model*	*x*	*-1.49555384*	*t*	*Grass Decay*
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
grass, semi-natural	*_model*		1.49555384	t	

Table A16: Grass Silage Storage.

Inputs	Category	Waste Flow	Amount	Unit	Provider
grass, semi-natural	*_model*		1	t	Grass Collection
Transport	transport/Road/Transformation		23.75	t*km	2020_SSP2_NPi_transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Carbon dioxide, biogenic	Emissions to air		5	kg	
grass, semi-natural	*_model*		0.9721	t	

Table A17: Anaerobic Digestion – Grass-CoAD

Inputs	Category	Waste Flow	Amount	Unit	Provider
Electricity	energy/Electricity country mix/High voltage/Market		25.2	MJ	2020_SSP2_NPi_electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U
grass, semi-natural	*_model*		249	kg	Silage storage, natural grass
Heat	energy/Heat/Others/Market		128.31	MJ	2020_SSP2_heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U
manure, liquid			751	kg	Manure Indoor Storage
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
biogas	E:Water supply; sewerage		53.88	m3	
Carbon dioxide, biogenic	Emissions to air		0.548	kg	
Digestate, GrassAD	*_model*	x	937.6	kg	Storage Grass-CoAD
manure FA	*_model*	*x*	*-751*	*kg*	*Manure Outdoor Storage*
Methane, biogenic	Emissions to air/low. pop.		0.251	kg	

Table A18: Storage Digestate Grass-CoAD.

Inputs	Category	Waste Flow	Amount	Unit	Provider
grass, semi-natural	*_model*		1	t	Grass Collection
Transport	transport/Road/Transformation		23.75	t*km	2020_SSP2_ transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Carbon dioxide, biogenic	Emissions to air		5	kg	
grass, semi-natural	*_model*		0.9721	t	

Table A19: Field Application Digestate Grass-CoAD.

Inputs	Category	Waste Flow	Amount	Unit	Provider
Digestate, GrassAD	_model	x	1000	kg	
inorganic nitrogen fertiliser, as N	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-3.06*	*kg*	*2020_SSP2_NPi_inorganic nitrogen fertiliser, as N {RER}| nutrient supply from calcium ammonium nitrate | Cut-off, U*
inorganic phosphorus fertiliser, as P2O5	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-0.568 * p2O5_P*	*kg*	*2020_SSP2_NPi_inorganic phosphorus fertiliser, as P2O5 {RER}| nutrient supply from diammonium phosphate | Cut-off, U*
inorganic potassium fertiliser, as K2O	*material/Chemicals/Fertilisers (inorganic)/Transformation*		*-2.769 * K2O_K*	*kg*	*2020_SSP2_NPi_inorganic potassium fertiliser, as K2O {RER}| nutrient supply from potassium chloride | Cut-off, U*
liquid manure spreading	processing/Agricultural/Transformation		1	m3	2020_SSP2_NPi_liquid manure spreading, by vacuum tanker {CH}| liquid manure spreading, by vacuum tanker | Cut-off, U

transport	transport/Road/Transformation		10-10*EMu	t*km	2020_SSP2_NPi_transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Ammonia	Emissions to air/low. pop.		(0.0157+0.705)*NH3_N	kg	
Carbon dioxide, bio-genic	Emissions to air		149.4	kg	
Dinitrogen monoxide	Emissions to air/low. pop.		(0.0437+0.00725)*N2O_N	kg	
Nitrate	Emissions to water/groundwater		(0.00150+0.2005)*NO3_N	kg	
Nitrogen oxides	Emissions to air/low. pop.		0.0044 * NO_N	kg	
Phosphate	Emissions to water		0.0166*PO4_P	kg	

Grass-Gasf

Table A20: Grass Drying for Gasification.

Inputs	Category	Waste Flow	Amount	Unit	Provider
electricity	energy/Electricity country mix/High voltage/Market		9.9	MJ	2020_SSP2_ high voltage {FI}| market for electricity, high voltage | Cut-off, U
grass, semi-natural			2.024	t	Grass Collection
heat	energy/Heat/Others/Market		250	MJ	heat, district or industrial, other than natural gas {Europe without Switzerland}| mar-ket for heat, district or industrial, other than natural gas | Cut-off, U
Transport	transport/Road/Transformation		200-200*Emu	t*km	2020_SSP2_ transport, tractor and trailer, agricultural {CH}| transport, tractor and trailer, agricultural | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
grass, semi-natural	*_model*		0.73891	t	

Table A21: Grass Gasification

Inputs	Category	Waste Flow	Amount	Unit	Provider
Electricity	energy/Electricity country mix/High voltage/Market		83	kWh	2020_SSP2_NPi_electricity, high voltage {FI}| market for electricity, high voltage | Cut-off, U
grass, semi-natural	_model		1000	kg	Grass Drying
heat	energy/Heat/Others/Market		150	MJ	2020_SSP2_NPi_district or industrial, other than natural gas {Europe without Switzer-land}| market for heat, district or industrial, other than natural gas | Cut-off, U
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
biochar	_model	x	12.1	kg	Biochar Field Application
Synthetic gas	_model		2367.1498	m3	

Table A22: Combustion Syngas CHP

Inputs	Category	Waste Flow	Amount	Unit	Provider
heat	*energy/Heat/Others/Market*		*-(14700 * heateff_syn-gas - 400) * 0.6 + 400*	*MJ*	*2020_SSP2_NPi_heat_ district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U*
Synthetic gas			2367.149758	m3	Grass Gasification
Outputs	**Category**	**Waste Flow**	**Amount**	**Unit**	**Provider**
Carbon monoxide, biogenic	Emissions to air/low. pop.		301	g	
Dinitrogen monoxide	Emissions to air/stratosphere + tropo-sphere		20	g	
electricity	energy/Electricity country mix/High voltage/Market		14700 * eleceff_syngas - 334.8	MJ	2020_SSP2_Npi_ high voltage {FI}| market for electricity, high voltage | Cut-off, U
fly ash	waste treatment/Incineration/Hazard-ous waste incineration/Market	x	0.6	kg	2020_SSP2_NPi_and scrubber sludge {Europe without Switzerland}| market for fly ash and scrubber sludge | Cut-off, U
Hydrochloric acid	Elementary flows/Emission to air/low population density		512	g	
Hydrogen chloride	Emissions to air		512	g	
Methane, biogenic	Emissions to air/low. pop.		7.2	g	
Nitrogen oxides	Emissions to air/low. pop.		855	g	
Particulate Matter, < 2.5 um	Emissions to air/low. pop.		0.7	g	
Particulate Matter, > 10 um	Emissions to air/low. pop.		28	g	
Particulate Matter, > 2.5 um and < 10um	Emissions to air/low. pop.		0.3	g	
Sulfur dioxide	Emissions to air/low. pop.		389	g	

II. LCIA Results

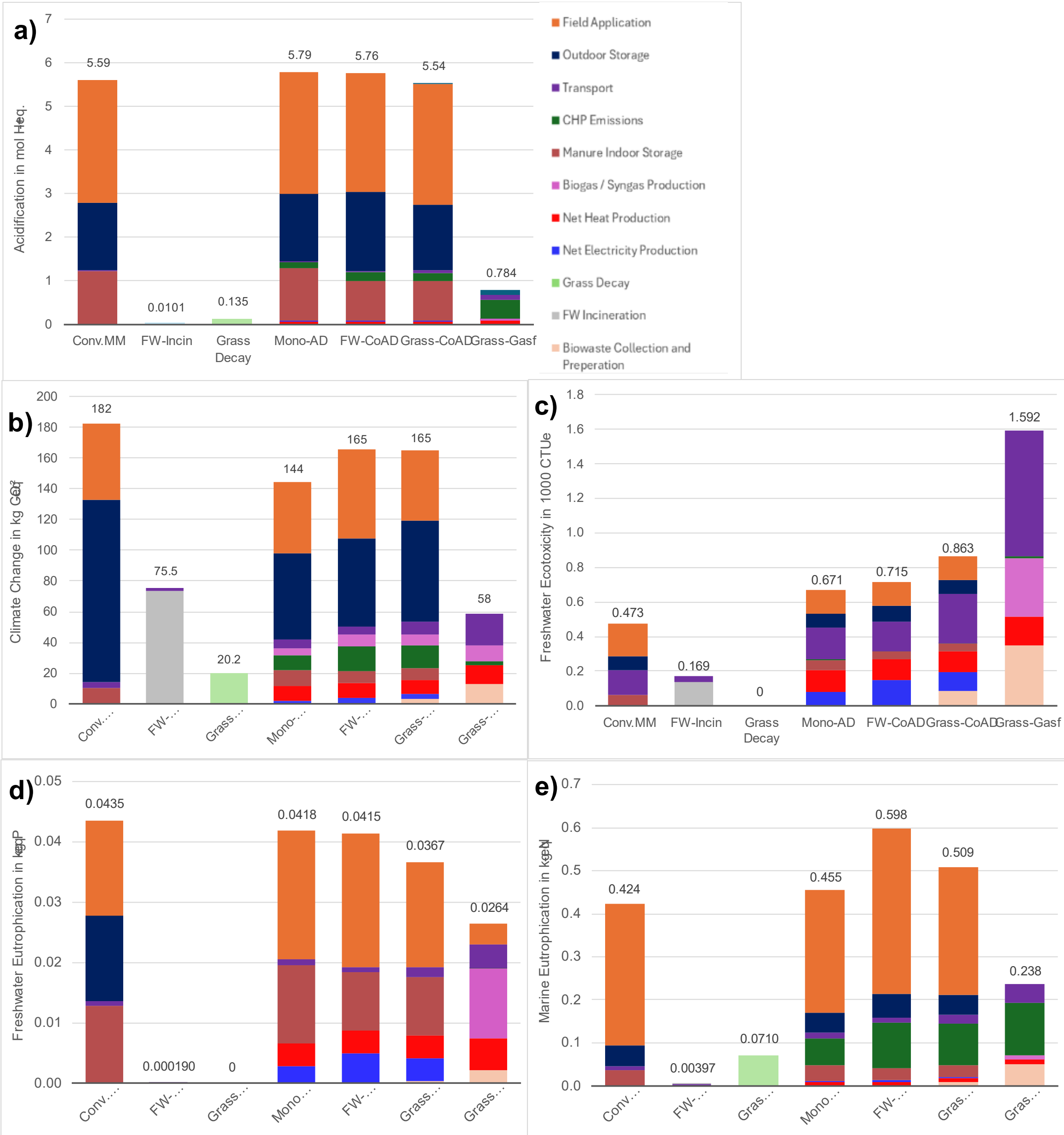

Figure A1: Complete overview of attributional impacts per 1 tonne of biowaste treatment.
Source: Own data and design.

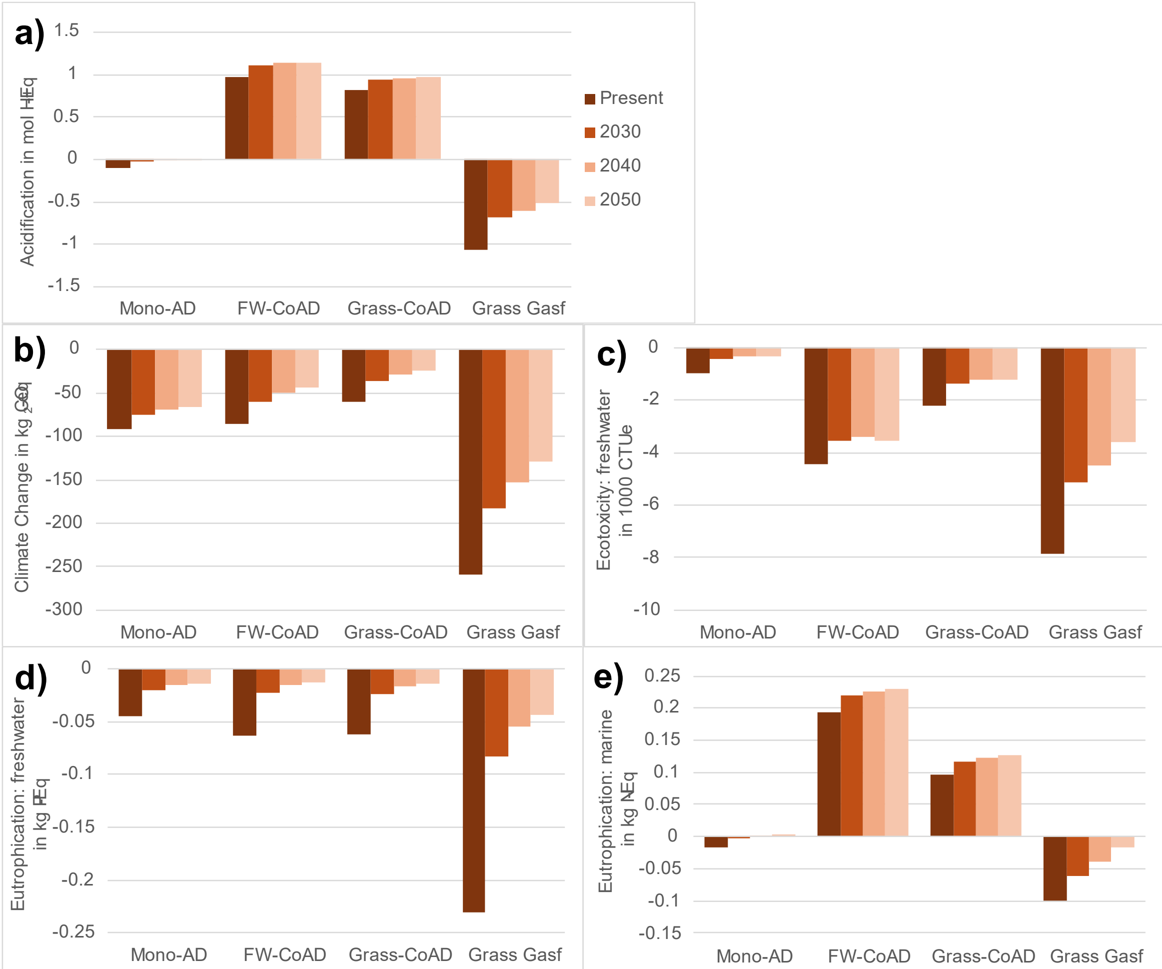

Figure A2: Prospective consequential impacts of all valorisation pathways in all four time frames.
Source: Own data and design.

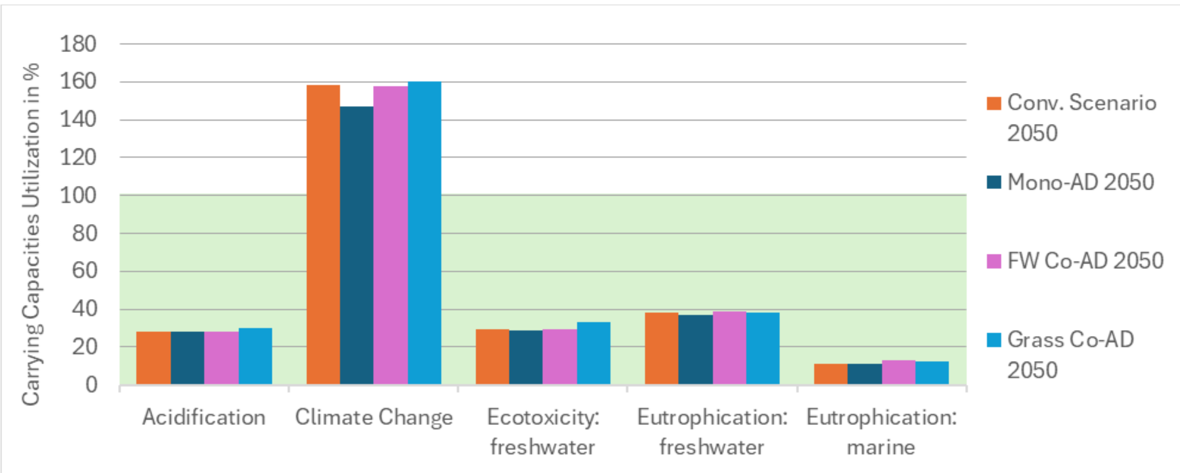

Figure A3: Impacts from the municipal development scenarios compared in the 2050 time frame, normalized by the carrying capacity per municipal inhabitant.
Respective carrying capacity as green background.
Source: Own data and design.

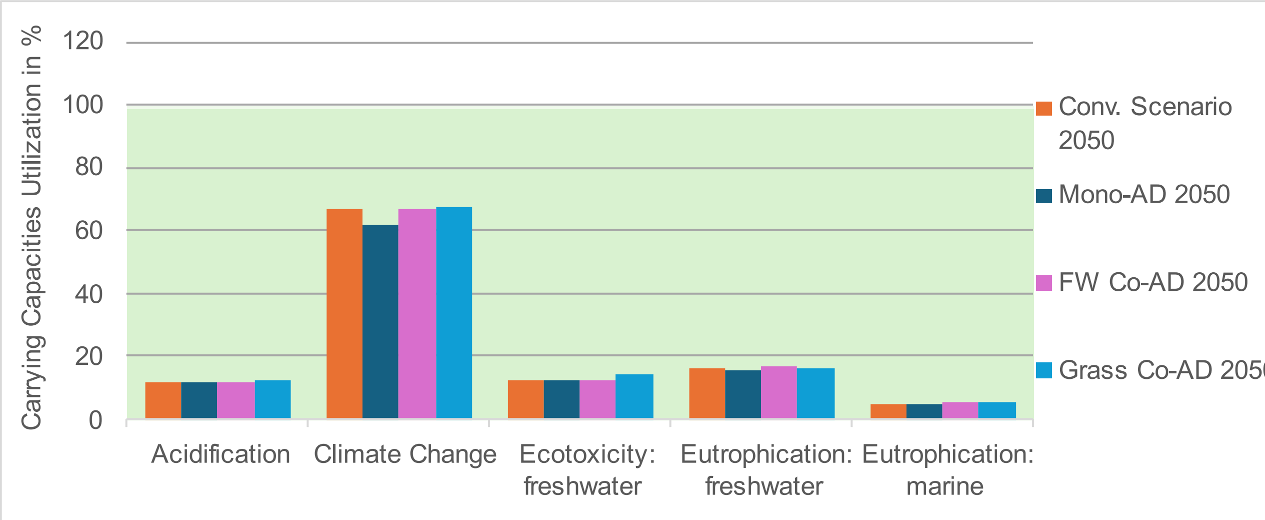

Figure A4: Impacts from the municipal development scenarios compared in the time 2050 frame, normalized by the carrying capacity per municipal land area.
Respective carrying capacity as green background.
Source: Own data and design.

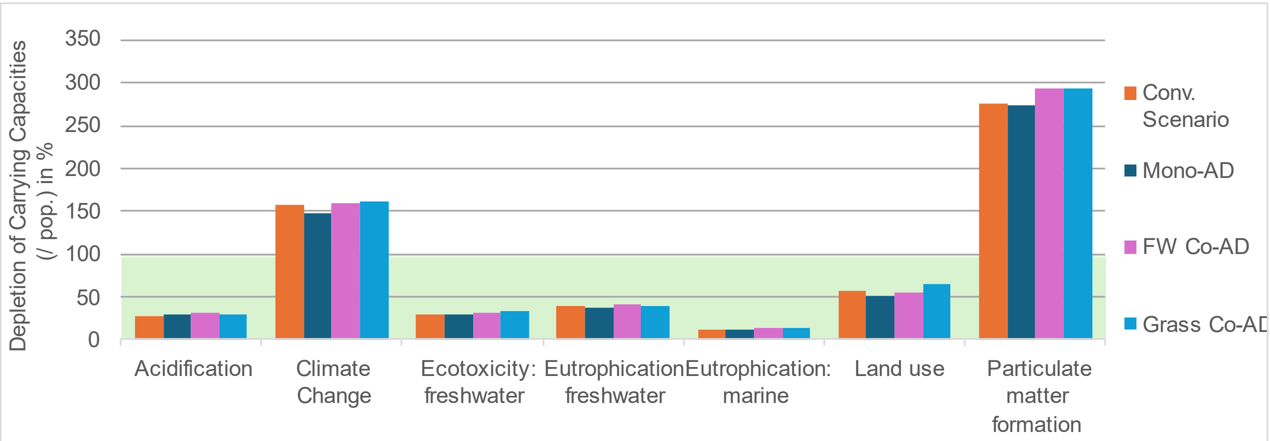

Figure A5: Impacts of municipal development scenarios in the time frame until 2050, normalized by the carrying capacity per municipal inhabitant.
Respective carrying capacity as green background.
Source: Own data and design.

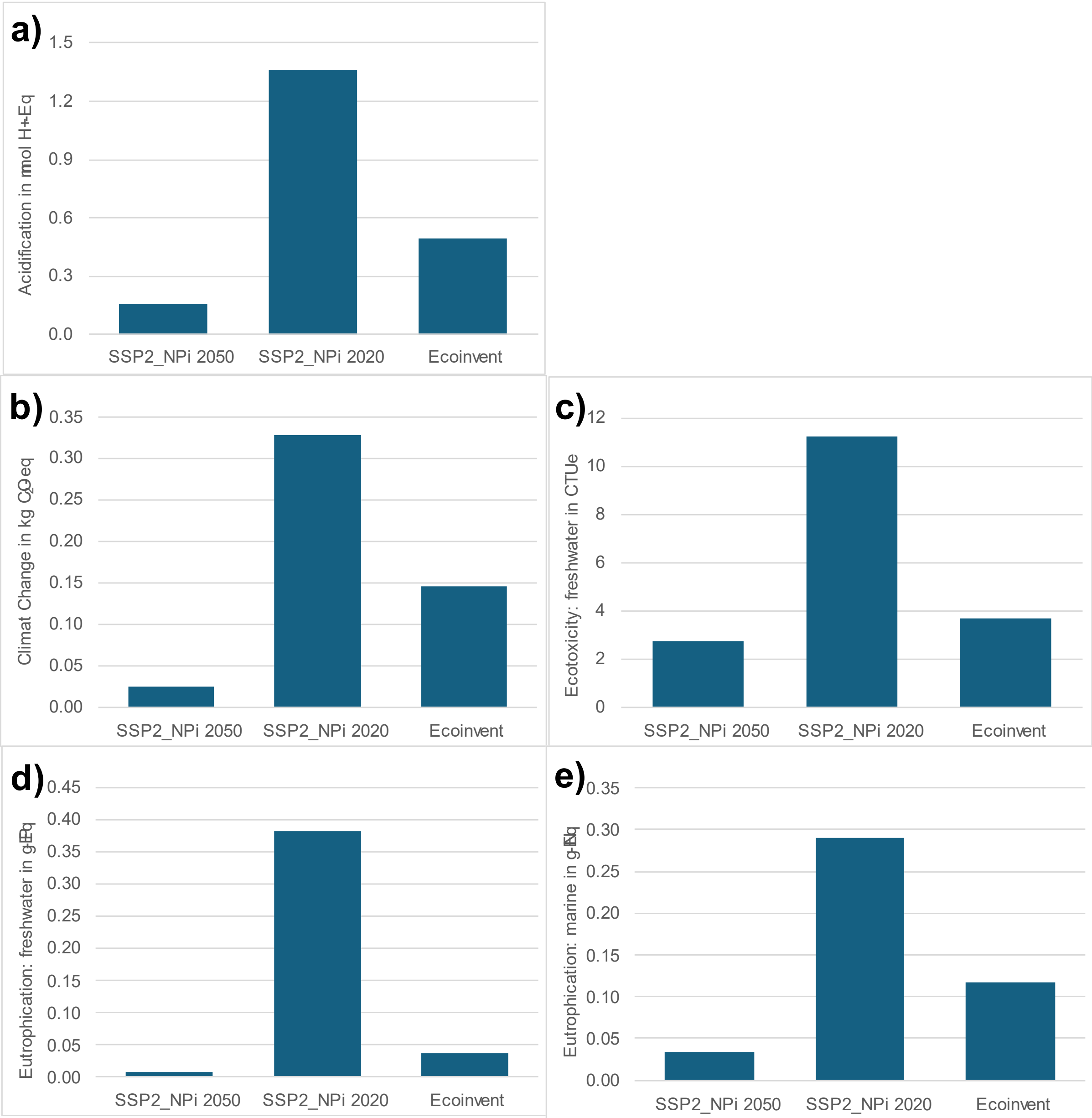

Figure A6: Impacts of 1 kWh of electricity in the Finish market mix in different databases: Two time frames of the SSP2_NPi scenario and Ecoinvent Cut-Off.