Acta Universitatis Agriculturae Sueciae

Doctoral Thesis No. 2025:89
Faculty of Natural Resources and Agricultural Sciences

Unlocking Untapped Food Resources

Environmental impacts of advancing resource efficiency in food systems via recovery of surplus and by-products

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SWEDISH UNIVERSITY
OF AGRICULTURAL
SCIENCES

DOCTORAL THESIS

Uppsala 2025

Acta Universitatis Agriculturae Sueciae
2025:89

Cover: Navigating recovery pathways for advancing resource efficiency in food systems (illustration: Louise Bartek)

ISSN 1652-6880
ISBN (print version) 978-91-8124-073-3
ISBN (electronic version) 978-91-8124-119-8
https://doi.org/10.54612/a.89fkd3er15

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Print: SLU Grafisk service, Uppsala 2025

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Abstract

Modern food systems face mounting pressure to reduce waste, enhance resource efficiency, and operate within planetary boundaries. Despite substantial efforts to curb losses and wastage of edible food resources, large volumes of underutilized surplus food and by-products are still generated throughout the food value chain, representing a vast untapped resource reservoir. This thesis explores the environmental implications of recovering these food resources and circulating them back into the food supply chain to support more sustainable, resilient, and circular food production.

Using a system-level life cycle assessment (LCA) framework, the research examines the potential environmental benefits and trade-offs of food resource recovery across different pathways and scales, focusing primarily on the Swedish and European contexts. The four studies cover strategies ranging from prevention and direct reuse to bioconversion, genetic engineering, and technological interventions, each representing distinct recovery pathways for recovering fat, protein, and carbohydrates for reuse in food or feed applications. The results demonstrate that when recovered resources replace conventional food and feed ingredients, considerable reductions in environmental impacts can be achieved, including reduced climate impact, lower nutrient pollution, less land use, and mitigated damage to ecosystems and biodiversity loss.

Collectively, the findings demonstrate that the environmental benefits of food resource recovery depend not only on the recovery process itself, but on how effectively surplus food and by-products can be reintegrated into the food system and thereby substitute food or feed resources. By linking environmental performance with technological readiness and system-level applicability, the thesis evaluates conditions under which recovery can most effectively advance resource efficiency. These insights provide a scientific foundation for navigating future development toward reconceptualizing otherwise wasted food into valuable resources for sustainable food systems.

Keywords: Recovery pathways, Life cycle assessment, Sustainable food systems, Resource efficiency, Food loss and waste prevention

Outnyttjad potential i livsmedelskedjan

Miljöpåverkan av ökad resurseffektivitet i livsmedelssystem genom återcirkulering av överskott och biprodukter

Dagens moderna livsmedelssystem står inför en kritisk utmaning i att minska matsvinn och öka resurseffektiviteten, utan att överskrida planetära gränser. Trots omfattande insatser för att minska livsmedelsförluster och matsvinn, genereras fortfarande stora mängder överskott och biprodukter längs hela livsmedelskedjan. Idag betraktas dessa resurser ofta som ett avfall, trots att de sannolikt rymmer en betydande, men ännu outnyttjad, potential att bidra till både en mer hållbar och resurseffektiv livsmedelsförsörjning.

Denna avhandling syftar till att kvantifiera den potentiella miljönyttan av att återcirkulera dessa outnyttjade resurser till livsmedelssystemet, med fokus på överskott och biprodukter. Genom en serie av fyra livscykelanalyser syftar arbetet även till att identifiera hur, och under vilka förutsättningar, olika strategier för återcirkulering kan bidra till ökad resurseffektivitet i Sverige och Europa. De fyra delstudierna inkluderar förebyggande åtgärder och re-distribution för minskat svinn, samt biokonvertering, genteknik, och teknologiska innovationer, som tillsammans möjliggör återcirkulering av värdefulla näringsämnen, inklusive fett, protein och kolhydrater. Dessa resurser kan i sin tur substituera primärproduktion av livsmedel eller foder genom att återintegreras i livsmedelssystemet. Resultaten visar att när återvunna resurser ersätter konventionella livsmedels- och foderråvaror kan betydande miljövinster erhållas, inklusive lägre klimatutsläpp, mindre föroreningar och markanvändning, samt reducerad skada på ekosystem.

Sammantaget visar resultaten att dessa miljömässiga fördelar inte bara beror på själva återcirkuleringsprocessen, utan även på hur effektivt överskottet och biprodukter kan återintegreras i livsmedelssystemet och vilka resurser de kan ersätta. Genom att utvärdera dess miljöpåverkan, begränsningar, och framtidspotential adresserar avhandlingen även hur, och under vilka förutsättningar, dessa strategier för återcirkulering kan bidra till ökad hållbarhet och resurseffektivitet. Arbetet bidrar därigenom till den växande kunskapsbasen som krävs för att främja hållbar innovation och utveckling av cirkulära livsmedelssystem.

Nyckelord: Resursåtervinning, Livscykelanalys, Hållbara livsmedels-system, Minskat matsvinn, Resurseffektivitet

Preface

There is nothing permanent, except change.

- *Heraclitus, a Greek Philosopher*

Dedication

To Hans, for your endless support and patience, and to my beloved son Leon, who gives life meaning and hope for the future.

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. Bartek, L., Strid, I., Henryson, K., Junne, S., Rasi, S. & Eriksson, M. (2021). *Life cycle assessment of fish oil substitute produced by microalgae using food waste*. Sustainable Production and Consumption, 27. https://doi.org/10.1016/j.spc.2021.04.033

II. Bartek, L., Sundin, N., Strid, I., Andersson, M., Hansson, P.-A. & Eriksson, M. (2022). *Environmental benefits of circular food systems: The case of upcycled protein recovered using genome edited potato*. Journal of Cleaner Production, 380. https://doi.org/10.1016/j.jclepro.2022.134887

III. Bartek, L., Sjölund, A., Brancoli, P., Cicatiello, C., Mesiranta, N., Närvänen, E., Scherhaufer, S., Strid, I., & Eriksson, M. (2025). *The power of prevention and valorisation – Environmental impacts of reducing surplus and waste of bakery products at retail.* Sustainable Production and Consumption, 55. https://doi.org/10.1016/j.spc.2025.01.013

IV. Bartek, L., Carlberg, H., Erichsen, M., Lalander, C., Guidini Lopes, L., Strid, I., Vidakovic, A., Wiklicky, V., Øverland, M. & Eriksson, M. (2025). *Closing the Rainbow-loop: Environmental impacts of using surplus food and upcycled resources in aquafeed* (submitted)

The contribution of Louise Bartek to the papers included in this thesis were:

I. Planned the paper in cooperation with the co-authors. Performed the data collection, impact assessment and analysis. Wrote the paper in cooperation with the co-authors.

II. Planned the paper together with the co-authors. Performed the data collection, impact assessment and analysis. Wrote the paper with support from the co-authors.

III. Planned the paper together with the co-authors. Performed the material flow analysis and data collection, impact assessment and analysis. Wrote the paper with input from the co-authors.

IV. Planned the paper with input from the co-authors. Performed the data collection and scenario development, impact assessment and analysis. Wrote the paper with input from the co-authors.

Papers produced but not included in the present thesis:

V. Eriksson, M., **Bartek, L.**, Löfkvist, K., Malefors, C. & Olsson, M.E. (2021). *Environmental Assessment of Upgrading Horticultural Side Streams - The Case of Unharvested Broccoli Leaves*. Sustainability, 13. https://doi.org/10.3390/su13105327

VI. Weber, L., **Bartek, L.**, Brancoli, P., Sjölund, A. & Eriksson, M. (2023). *Climate change impact of food distribution: The case of reverse logistics for bread in Sweden.* Sustainable Production and Consumption, 36. https://doi.org/10.1016/j.spc.2023.01.018

VII. Sundin, N., **Bartek, L.**, Persson Osowski, C., Strid, I. & Eriksson, M. (2023). *Sustainability assessment of surplus food donation: A transfer system generating environmental, economic, and social values*. Sustainable Production and Consumption, 38. https://doi.org/10.1016/j.spc.2023.03.022

VIII. Hildersten, S., **Bartek, L.**, Brancoli, P., Eriksson, M., Karlsson Potter, H. & Strid, I. (2025). *Mapping the Climate Impact of Rye Bread Production in Sweden: Insights into Cultivation, Packaging, and Surplus Management for Sustainable Food Systems.* Frontiers in Sustainable Food Systems, 9. https://doi.org/10.3389/fsufs.2025.1528862

IX. **Bartek, L.**, Bergquist, D., Garcia-Caro, D., Malefors, C. & Eriksson, M. (2025). *Sustainability beyond buildings: Assessing environmental impacts of Swedish urban life using LCA and emergy indicators.* Cleaner Environmental Systems, 19. https://doi.org/10.1016/j.cesys.2025.100343

X. Eriksson, M., **Bartek, L.**, Sturén, F., Christensen, J., Cicatiello, C., Giordano, C., Malefors, C., Pasanen, S., Sjölund, A., Strid, I.V., Sundin, N. & Brancoli, P. (2025). *From surplus to sustainability: The role of legislation in reducing climate impact from Swedish bread waste.* Current Research in Environmental Sustainability, 10. https://doi.org/10.1016/j.crsust.2025.100301.

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Abbreviations

CO_2eq	Carbon dioxide equivalent
EC	European Commission
EU	European Union
FAO	Food and Agricultural Organisation of the United Nations
LCA	Life cycle assessment
MFA	Material flow analysis
UN	United Nations
SDG	Sustainable development goal

1. Introduction

Food is an essential part of our lives, not only providing the nutrients we need to nourish our bodies but also shaping our cultures and sustaining our survival. However, the growing demand for natural resources as the global population increases, coupled with substantial inefficiencies in the production and management of food resources, represents one of the most pressing challenges of our time (Springmann et al. 2018). In fact, our modern food systems are identified as key drivers of environmental degradation, already contributing to the transgression of six out of nine planetary boundaries (Richardson et al. 2023), accelerating climate change, biodiversity loss, and depletion of natural resources (Ritchie et al. 2022). About one-third of the world's food is lost or wasted along the supply chain (United Nations Environment Programme 2024), whereas on an EU level, about 59 million tonnes of food resources are never used according to their intended purpose (Eurostat 2024). In Sweden alone, over one million tonnes of solid food are wasted annually, much of which ends up in energy production rather than on our plates (Johansson 2021; Hultén et al. 2024). This implies a drastic wastage of available resources when food is not used according to its highest potential value. In turn, this undermines food system resilience and sustainable development (Bajželj et al. 2020). As the global population is predicted to approach 10 billion by 2050, closing the estimated food gap of 56% (Searchinger et al. 2018) becomes increasingly urgent.

In 2019, the *EAT-Lancet Commission* highlighted the need to transform global food systems to deliver sustainable diets within planetary boundaries (Willett et al. 2019). Its 2025 update reinforces this vision, emphasizing resilience and regionally adapted solutions, further stressing the importance of advancing circular and resource-efficient approaches to ensure food security within environmental limits (European Parliament 2025; Rockström et al. 2025). A considerable challenge within this field lies in the extensive losses and untapped resource potential of food systems, including large quantities of surplus and by-products arising during the earlier stages of the supply chain. This emphasises the urgent need for a systematic shift toward circular food systems, where food is also produced using resources currently wasted. To guide such efforts, the food waste hierarchy could provide a structured framework for prioritising how food resources should be managed based on their highest potential value and lowest environmental impact (De Laurentiis et al. 2024).

To address this challenge, a growing number of innovative approaches are emerging to recover and upcycle surplus and by-products originating from food resources (Schieber 2017; Moreno-González & Ottens 2021). Some examples include process optimisation, bioconversion, and technologies for valorising food. When effectively integrated, such pathways could reduce environmental pressures by substituting raw materials, avoiding the generation and management of waste, and support alignment of production with actual consumption needs, while simultaneously enhancing resilience and circularity within food and feed systems (European Commission 2023). However, identifying how, and under what conditions, different pathways for recovering surplus or by-products entail reduced environmental impacts remains insufficiently studied (Thorsen et al. 2025). Filling this knowledge gap, by assessing the environmental impacts of reducing food waste via recovery of valuable food resources, is therefore of utmost importance for advancing more sustainable, resilient, and resource-efficient food systems.

This thesis contributes to closing these knowledge gaps by assessing the environmental impacts of recovering and reusing untapped food resources, via four case studies in Swedish and European contexts. By applying environmental life cycle assessment (LCA) to identify and navigate the benefits and potential trade-offs of each recovery pathway, this work examines whether the environmental gains of added resource efficiency outweigh the impacts introduced through food resource recovery. Via this lens, the thesis contributes to the growing body of knowledge needed to guide the transition toward sustainable, more resource-efficient food systems, where no valuable resource is wasted, and every by-product has a purpose.

2. Aim, objectives, and thesis structure

2.1 Aim and objectives

The objective of this thesis is to evaluate the environmental impacts associated with food waste reduction by recovering surplus and by-products in Sweden and Europe, using different pathways to circulate resources back to the food supply chain. The research investigates actions spanning the top three levels of priority outlined in the *food waste hierarchy*: prevention, direct reuse in food, and reuse in animal feed, aiming to recover resources according to their highest potential value. To achieve this objective, the specific goals of the thesis are:

- Assessing the environmental impacts of recovering essential nutrients from surplus food and by-products via three main recovery pathways, including bioconversion, genetic engineering, and technology interventions, representing different strategies for increased resource efficiency in the food supply chain (*Papers I–III*).

- Assessing the environmental impacts and system-level implications of substituting conventional aquafeed ingredients with recovered surplus and by-products, building on the findings from *Papers I–III* to evaluate their practical application in a real product system. This assessment provides insights into how such substitutions can support resource efficiency and reduce environmental pressures (*Paper IV*).

2.2 Thesis structure

The central theme of this work is assessing how, and under what conditions, surplus food and by-products can become a tool for reducing food waste and supporting a sustainable supply of food available for consumption. With this objective, this thesis aims to answer the key question: how environmentally beneficial is it to produce food using resources recovered from surplus or by-products? Especially focusing on how available food resources can be reprocessed to recover valuable macronutrients, and how these can be circulated back to the food system. The thesis is structured to assess pathways for recovering food resources via the following levels of priority (Figure 1): reuse and low-value valorisation (*Papers I–II*), prevention and high-value valorisation (*Paper III*), and reuse in animal feed (*Paper IV*).

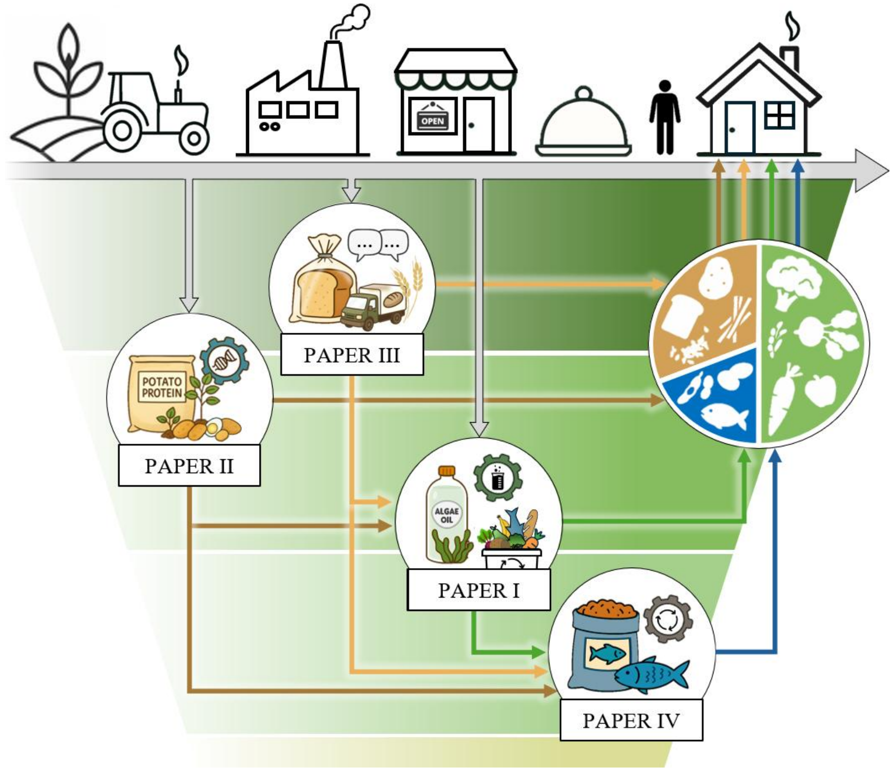

Figure 1. Structure of the thesis, illustrating the assessed recovery pathways for food resources arising from primary production, the food industry, and the retail level.

Papers I–III focus on the environmental impact of producing fat, protein, and carbohydrates via recovered resources arising from primary production, the food industry, up to retail level, and the service sector, via bioconversion (*Paper I*), genetic engineering (*Paper II*), and technology interventions (*Paper III*). In the final papers, the approach is expanded to include a systems perspective. Firstly, by assessing *direct reuse* via prevention and high-value valorisation as food ingredients (*Paper III*), and secondly by assessing the environmental impacts of directing recovered food resources toward *reuse in animal feed* to support domestic aquaculture production (*Paper IV*).

3. Background

Our modern food value chain typically follows a linear model comprising six main stages, including primary production, food industry, distribution, retail and service sectors, household, and waste management (Figure 2). Primary production refers to the stage during which raw materials from plants and animals are produced, including activities such as agriculture and aquaculture. The food industry stage involves processing and manufacturing of these raw materials into food products, such as milling grains, trimming vegetables, ingredient production, and baking. Distribution encompasses the transportation and logistics required to move food products from primary production and processing sites to points of sale. Retail stage comprises wholesale and stores in which products are made available to consumers, while the service sector includes actors that cook and sell meals, such as restaurants, food services, and school canteens. At the household level, food is stored, prepared, and consumed in private homes. Finally, the waste management stage comprises the collection, treatment, and disposal of lost or wasted food resources generated across all previous stages.

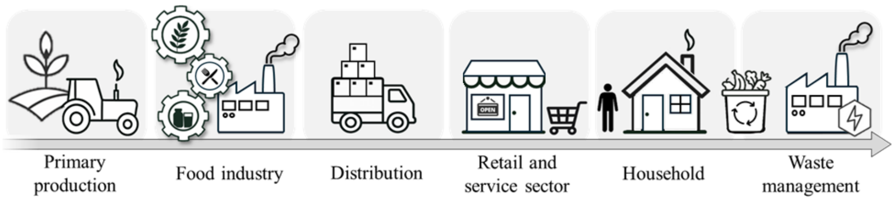

Figure 2. Illustration of the food value chain.

The underlying linearity of the food system generally infers that discarded or underutilized food resources are removed from the food chain and directed toward energy production rather than recovered as food or feed. This loss reflects a system optimized for throughput, rather than retention of nutritional and economic value. As food moves through this chain, it is increasingly treated as a commodity, being priced and managed according to industrial logics (Yontar 2025). This framing has profound implications for how food is valued, managed, and ultimately wasted in modern economies.

3.1 Food as an industry

At each stage of the food value chain, substantial amounts of food resources are lost or wasted. This resource inefficiency is further compounded by how food is conceptualized and valued. While food is fundamentally a source of essential nutrients, its role in modern economies is often reduced to that of a tradable commodity. Today, food is primarily regarded through the lens of an industry product, where its production and management are driven by efficiency, productivity, and profit margins (Khanna et al. 2024). This framing, while central to the development of today's globalized food systems, tends to sideline the deeper social, nutritional, and ecological values of food. As Gordon et al. (2022) note, treating food predominantly as a commodity has reinforced a system where its production and consumption are governed by market logics rather than the imperatives of sustainability, justice, or health. This industrial perspective can lead to short-term gains but often at the expense of long-term environmental resilience and societal well-being. Although this shift has in many ways served modern lifestyles and sustainable development, it has also altered dietary habits and introduced new challenges related to industrialized food systems, such as convenience, overproduction, and disposability (Wood et al. 2019; Habib et al. 2025).

This detachment contributes to systemic inefficiencies and substantial losses throughout the food chain. When food is not valued as a precious resource, wastage increases, from production and processing to retail and households (Chauhan et al. 2021; Harwatt et al. 2024). Moreover, this undervaluation undermines efforts to build self-sufficiency and resilience in national food systems (Bajželj et al. 2020), especially when untapped resources are directed toward energy production rather than human consumption or feed application (Muscat et al. 2020; Johansson 2021). Breaking this pattern, both in the European and global contexts, requires new approaches that reframe food as not merely a product of industry, but as a vital and valuable resource.

3.1.1 Why are food resources wasted?

Despite global efforts to reduce the wastage of food resources (FAO 2018a), staggering amounts are lost early in the supply chain, long before reaching the consumer level. At the primary production stage, several factors contribute to these losses (Parfitt et al. 2021; Stoica et al. 2022). In Sweden, for example, it is estimated that 20-30% of root vegetables, such as carrots and potatoes, are left in the fields or discarded after harvest (Lindow et al. 2024). Similar patterns are observed across the broader European context (Hartikainen et al. 2017; Scherhaufer et al. 2018), often due to strict cosmetic standards, weather damage, pest attacks, or production exceeding market demand (Porter et al. 2018; Clausen et al. 2025; Philippidis et al. 2025). This aspect has also been highlighted for crops like broccoli, where large portions of the plant are edible but not harvested since not considered marketable

(Eriksson et al. 2021). Extreme rainfall or early frosts can severely damage crops before harvest, leading to downgraded quality or completely lost yields. In other regions, droughts are a persistent threat to food security (Saleem et al. 2025). Moreover, inefficient harvesting practices, such as picking too early or mechanical damage, result in considerable avoidable losses (Joensuu et al. 2021). In terrestrial animal farming, losses often arise from reproductive failures, disease outbreaks, injuries or welfare issues, and climate stressors (Godde et al. 2021; FAO 2023a). Similar aspects are also drivers for losses and waste in aquaculture, alongside factors such as feeding inefficiencies, handling stress during rearing, physical damage, and poor water conditions (Lindow et al. 2021; Singh et al. 2024).

During the food industry stage, factors such as inefficient handling, improper storage, and removal of edible parts are common causes (Chauhan et al. 2021). For instance, in juice or chutney production, large volumes of pulp are discarded despite being rich in fiber and nutrients (Spångberg & Eriksson 2016), while peelings and trimmings from plant-based products (Panouillé et al. 2007; Hussain et al. 2020) alongside by-products and downgraded materials from animal and fish processing, such as bones, offal, blood, and trimmings, also represent considerable losses (Gustavsson et al. 2011). Additional examples are residual dough or misshapen products from bakeries (Hildersten et al. 2025), and removed protein fractions during industrial potato starch production (Godard et al. 2012). Substantial amounts are also discarded due to overproduction (Messner et al. 2021), suboptimal handling and storing (Rao et al. 2021), and quality requirements (Clausen et al. 2025).

At the retail level, losses of perishable items like leafy greens, soft fruits, and baked goods remain notably high (Trento et al. 2021). Fruits and vegetables are especially vulnerable to bruising, dehydration, and cosmetic rejection. Overordering to ensure full shelves often leads to excess stock, while poor stock rotation or inventory mismatches result in wastage of food products (Olabode et al. 2025). For instance, surplus bread or dairy products often remain unsold because of their short shelf life, despite being edible for several days beyond their “best before” date (Secondi 2019). Additionally, consumer behaviour, such as selecting only the most visually perfect produce, means that slightly blemished but perfectly edible items are left behind and eventually discarded (Lebersorger & Schneider 2014). Physical damage, such as torn packaging, alongside potential mismatch between pre-packaged portions and consumer preference, also contributes to waste at retail (Svanes et al. 2019). For aquaculture and animal-based products, additional factors such as high perishability, temperature control failures, and strict visual or freshness standards also contribute to wastage of edible food (Eriksson et al. 2016; de Moraes et al. 2020; Nethra et al. 2023). Meanwhile, factors such as menu variety, portion oversizing, plate waste, and overproduction due to unpredictable customer demand have all been identified as drivers of food waste at service levels (Malefors et al. 2019; 2024; Dhir et al. 2020). Together, these examples illustrate how wastage of food is driven by a combination of market-driven practices and supply chain mismanagement, across every stage from farm to retail shelf and restaurant table. Moving forward, targeted strategies must recover these resources.

3.1.2 How much food resources are wasted?

The issue of wasting still edible food has existed throughout history, although the causes driving it have changed considerably over the years. Wastage occurring during the earlier stages of the supply chain was, back in the days, predominantly caused by uncontrollable factors like weather or lack of preservation technology (Habib et al. 2025). While these factors still play a vital role, they are nowadays accompanied by a set of modern challenges, e.g., strict cosmetic standards and linear food supply chains. Although the vast majority of food waste generated in the EU arises at the household level (54%), the remaining 46% wasted annually occurs during the earlier stages in the food supply chain (Soloha & Dace 2025). On a Swedish level, roughly 123 kg per capita is wasted, representing a total of 1.2 million tonnes of solid food (Eurostat 2024). These estimates are based on the definitions of food waste by Hultén et al. (2024), including both edible and inedible parts of the discarded food. Of this, about 0.64 million tonnes arise during primary production (7%), food industry (25%), retail and distribution (8%), alongside the service sector (14%), respectively (Figure 3).

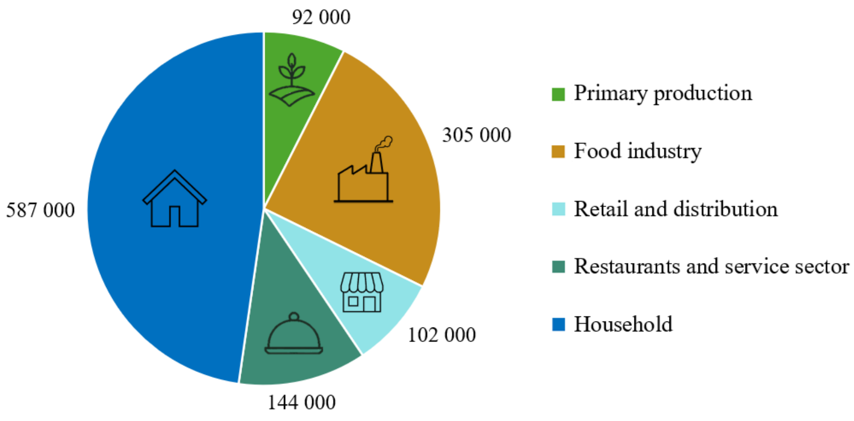

Figure 3. Wasted food resources by sector in Sweden 2022, expressed per tonnes.

Although these resources are produced with the intended purpose of human consumption, they are primarily directed toward energy recovery rather than being recovered for food or feed applications (Johansson 2021). However, a considerable research gap persists regarding the extent to which food is lost or wasted during these stages (Eriksson et al. 2019; Hoehn et al. 2023), as well as how these resources are managed and to what extent they could be recovered. This was also voiced by Lindow et al. (2024), stressing that actual wasted food quantities are likely substantially higher than current estimates.

This persistent research gap has previously been attributed to multiple factors, ranging from inconsistent definitions (Teigiserova et al. 2020) and a lack of consistency in quantification methods (Caldeira et al. 2019). As highlighted by Siddique et al. (2024), the limited scientific coverage on feasible, system-level recovery pathways for food resources also contributes to the challenge. Consequently, identifying the actual volumes of recoverable food resources and determining the most environmentally preferable recovery pathways for different food streams remains complex. Understanding these limitations is essential, as the environmental impacts of food are strongly influenced not only by production and consumption but also by the fate of resources lost along the supply chain.

3.2 Environmental aspects of food

Modern food systems are one of the key drivers of environmental impacts globally. As highlighted by Ritchie et al. (2022), agriculture is responsible for roughly 25% of the global greenhouse gas emissions and 70% of the global freshwater withdrawals. The environmental impact of food production is also massive with respect to land use, as half of the world's habitable land is currently used for agriculture (Ritchie & Roser 2019). Agricultural runoff, rich in nitrogen and phosphorus, has also been linked to considerable eutrophication and loss of aquatic biodiversity (Jwaideh et al. 2022). This contributes to a drastic environmental impact, while also inflicting immense pressure on natural resources. The expansion of food production to feed the growing population has further aggravated habitat degradation and deforestation (FAO 2018b), which are direct drivers of biodiversity loss (Crenna et al. 2019). Aquaculture, comprising farming of aquatic organisms such as fish and algae, has been increasingly promoted as a sustainable food source, but this sector also raises environmental concerns, including nutrient emissions, feed resource demands, resource use, and ecological impacts (FAO 2024; Cottrell et al. 2025; Deville et al. 2025).

Among dietary macronutrients, fat and protein contribute disproportionately to environmental impacts, reflecting the higher land, energy, and resource demands involved in their production and processing (Poore & Nemecek 2018). A large share of the environmental impacts of food can be traced to animal-based food production, which has consistently been shown to be more resource-intensive compared to plant-based alternatives (Detzel et al. 2021; Karlsson Potter & Röös 2021). In comparison, the production of meat, fish, dairy, and eggs generally requires more land, water, and energy inputs (Rysselberge & Röös 2021), while also requiring inputs in terms of feed. In turn, this creates a new set of challenges related to food-feed competition (Muscat et al. 2020), which becomes particularly problematic with respect to feeding a growing population with nutritious and safe food without further jeopardizing the already overstretched planetary boundaries. Gerten et al. (2020) highlighted dietary changes, favouring plant-based resources in food and feed, alongside food waste reduction as key prerequisites to ensure the future sustainability of our food system.

In Sweden, the average dietary climate impact was estimated by Hallström et al. (2021) at approximately 2.0 tonnes CO_2eq per person annually, with food loss and waste contributing to around 18% of this footprint. For comparison, this is nearly three times higher than the limit of 0.7 tonnes CO_2eq per capita proposed by Moberg et al. (2020) for sustainable global food systems. These figures underscore the urgent need for transformative changes in production and consumption patterns within the Swedish and broader European context, to alleviate environmental pressures, increase resource efficiency, and align food systems with sustainability targets (de los Mozos et al. 2020). Redesigning our food systems around circularity principles could substantially reduce these impacts (van Zanten et al. 2023).

3.3 Transforming the food system

Many global challenges of today, clearly outlined in the United Nations' 17 Sustainable Development Goals (SDGs), require urgent actions that drive the change needed to ensure a sustainable, fair, and prosperous future for people and the planet (Rockström et al. 2020; FAO 2023b). Due to the complexity of these challenges, many of which are deeply interconnected, a range of targeted strategies applied with a collaborative effort across various sectors and disciplines is needed to truly transform our food systems. This transformation involves rethinking what is traditionally regarded as "waste" and instead recognizing and recovering surplus food and by-products according to their inherent value (Philippidis et al. 2025).

3.3.1 Legislation and policy guidelines

Both the SDG and planetary boundaries are aimed at ensuring long-term global sustainability, accounting for the interconnectedness between human well-being and the stability of Earth's natural systems (Rockström et al. 2023). Although having different perspectives, these frameworks can complement each other, where SDGs can guide social, environmental, and economic progress, while respecting the planetary boundaries ensures that such progress doesn't exceed the Earth's ecological limits. This thesis focuses on two key SDGs: *Responsible Consumption and Production* (SDG 12), with particular emphasis on target 12.3 of halving global food waste per capita, alongside *Climate Action* (SDG 13).

While all SDGs are critical for building a sustainable future, SDG 13 is often seen as the most pressing issue (FAO 2023b:13). This target aims to keep greenhouse gas concentrations within safe limits and prevent global temperatures from rising beyond critical thresholds, also linked to the planetary boundary of climate change. Scientists warn that current trends are heading toward nearly 3°C of global warming, far exceeding the 1.5°C limit set in the 2015 *Paris Agreement* (United Nations 2016). To curb this trajectory, growing emphasis is placed on systemic transformations of the global food system (FAO 2025; Rockström et al. 2025). Such efforts include reducing food waste through improved resource recovery (United Nations Environment Programme 2024) and enhancing food system resilience through the sustainable use of surplus and by-products (Springmann et al. 2018; European Commission 2023). These actions are also tightly linked to SDG 12, addressing resource overuse and unsustainable production patterns, which in turn also aggravate pressures on planetary boundaries such as land-system change and nutrient pollution (Gerten et al. 2020; Richardson et al. 2023). Within this goal, target 12.3 specifies cutting global food waste in half by 2030, alongside incorporating food loss into a circular bioeconomy by 2050. Aligning with this goal, the European Commission (2025), including all member states, has committed to halving food waste per capita at the retail and consumer levels by 2030. In Sweden, the goal is to cut food waste by at least 20% per capita by 2025, while ensuring that more food reaches retailers and consumers (the Swedish Environmental Protection Agency 2024).

Achieving these global goals and targets, however, requires more than recognition; it demands actionable strategies grounded in research and innovation. In the *Action plan for food loss and waste*, Swedish Food Agency et al. (2025) emphasized that substantial and urgent efforts are required to meet these targets, including facilitated resource recovery, collaboration across sectors, and leveraging the flexibility of food legislation. By examining the environmental consequences of food waste and evaluating pathways for recovery and reuse, research can guide decision-makers toward interventions that are both effective and synergistic (McConville et al. 2015).

However, the latest United Nations (2025) report indicates regression towards achieving target 12.3. To accelerate the progress toward the fulfilment of this goal, legally binding food waste reduction targets were adopted under Directive (EU) 2025/1892 in the revised *Waste Framework Directive* (European Parliament 2025). Given the regression towards meeting the SDGs and the forthcoming legal reduction targets, the urgent need to assess how, and under what conditions, resource recovery pathways can effectively contribute to these objectives becomes increasingly evident.

3.3.2 Definitions of food resources

Reaching a common understanding of what qualifies as food loss or food waste remains a challenge (Leverenz et al. 2021), alongside arriving at a common methodology to quantify these losses and waste (European Union 2013). As an example, the term *by-product* is often used interchangeably with *side stream* when describing the generation of various food losses, although only the former is legally defined in the Waste Directive. In contrast to by-products, the term *side stream* often includes a broader range of resources, such as by-products and solid waste, usually produced without a clear market value or intended use. Similarly, the term *food resource* is not covered by a legal definition but is rather a comprehensive concept, including all available sources of the legally defined term *food*, alongside the systems for managing them (CRS 2014). Additionally, the term *resource-efficient* often used to describe the desired outcome of food system transformations, also lacks a legal definition, but is often used to describe systems aiming to produce and distribute food using fewer resources, less waste, and limited environmental impact (European Commission 2023). Although this ongoing debate is important, and consistent distinctions between surplus food and waste is crucial to implement efficient recovery pathways (Papargyropoulou et al. 2014), the focus should not be solely on definitions. Rather, focus should be placed on recognizing the inherent value of food and prioritising actions that allow these resources to be used according to their highest potential value. This thesis adopts the FAO (2019) definitions on food loss and food waste, as it accounts for where the resource is generated, while also aligning with the SDGs. Definitions of different food resources are presented in Table 1, alongside a description of terms not covered by a common definition.

Table 1. Definitions and descriptions of terms related to food resources.

Term	Definition, legally defined
Food (EC 2002)	Any substance or product, whether processed, partially processed, or unprocessed (raw), intended to be, or reasonably expected to be, ingested by humans. Often used to describe food resources. Legally defined.
By-product (EC 2008)	Quantity of food resources produced in addition to the main product, which can be reused under specific conditions without being classified as waste. Intentionally created, usually with a market value. Legally defined.
Food loss (FAO 2019)	A decrease in food quantity or quality, arising due to actions and decisions at production, post-harvest, and processing stages, excluding retail, service sector, and consumers. *Quantitative losses* refer to a decrease in the mass of food destined for human consumption, while *qualitative losses* refer to a decrease in food attributes or nutritional value that limits intended use. Legally defined.
Food waste (FAO 2019)	A decrease in the quantity or quality of food resulting from decisions and actions by retailers, food services, and consumers. *Quantitative waste* refers to a physical decrease in food mass, while *qualitative waste* refers to a decrease in food attributes or nutritional value that reduces the value related to intended use. Legally defined.
Term	**Description, not legally defined**
Surplus food (EC 2017)	Quantity of produced food resources exceeding the demand (including finished products, partly formulated products, and food ingredients) arising at any stage of the food value chain. Not legally defined. *Examples: overproduction or inefficient resource use, alongside products deemed below standard and undesirable parts such as peels or trimmings.*
Wasted food (EPA 2023)	Quantity of food resources not utilized for their intended purpose and instead diverted through recovery pathways, such as donations, animal feed, anaerobic digestion, and incineration. Captures the loss of resources with maintained value. Not legally defined. *Examples: unsold food items, plate waste, leftovers, excess food, surplus, and by-products.*

3.3.3 The food waste hierarchy

The *Waste Framework Directive* has been part of EU policy since 1975. After its revision in 2018, all Member States are required to reduce food waste generated during all stages of the supply chain, according to the level of priority outlined in the *Food Use and Waste hierarchy* (European Commission 2020), generally attributed to the work of Papargyropoulou et al. (2014). Prevention and redistribution of surplus is placed at the highest priority, followed by direct reuse as a food product or ingredient in food production, and thirdly, reuse in animal feed. Building on this work, Teigiserova et al. (2020) explored an adaptation to this framework, in which they accounted for the nutritional and economic value of the food resources, alongside a clear distinction between priorities for surplus resources and food waste. Similarly, Moshtaghian et al. (2021) added upcycled food production as a management action to their framework. Building on these perspectives, Figure 4 illustrates the pathways for recovery assessed in this work.

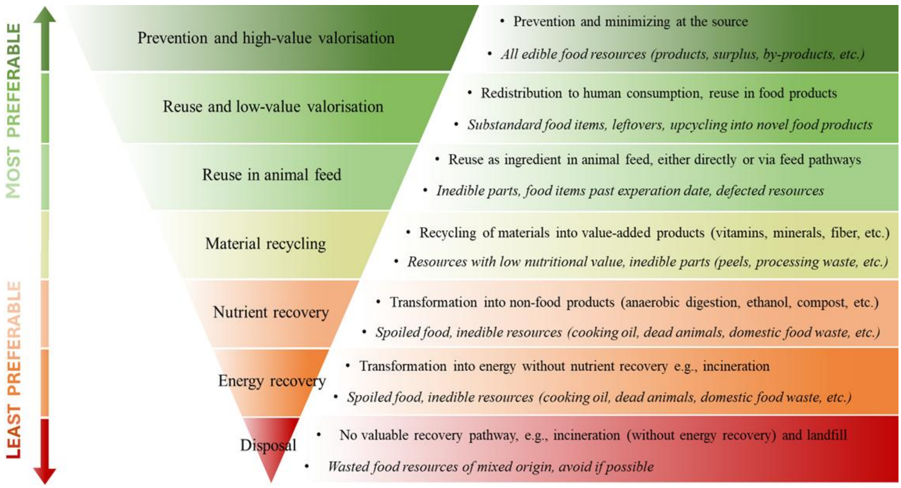

Figure 4. Food waste hierarchy framework used in this work, illustrating the levels of priority, recovery pathways, management strategies, and food resources treated.

Today, similar frameworks have been adopted by many policymakers worldwide (Giordano et al. 2020). Although the interpretation and classification of applied measures can differ, the joint consensus suggests prioritizing prevention and moving downward only when higher options are not feasible. Redistribution of surplus food, e.g., via donations, has become a key strategy to address food insecurity and wastage of edible food in many high-income countries (Sundin et al. 2023), with the revised waste framework directive further encouraging food donation and redistribution as a means of preventing food waste (European Parliament 2025).

However, this order of priority is not absolute. Instead, research suggests that hierarchy frameworks should be applied with flexibility, considering the specific value and environmental trade-offs associated with each resource (Moshtaghian et al. 2021; Parsa et al. 2023). In certain contexts, such as when redistribution or reprocessing requires substantial energy inputs or involves safety limitations, alternative recovery pathways, such as reuse in feed use, may provide greater overall sustainability benefits. Moreover, the levels of priority are not always feasible with respect to legal or regulatory barriers, such as food safety concerns and liability issues. As an example, the Regulation (EC) No 1069/2009 on animal by-products defines food waste as containing both animal and plant-based components, which means that even source-separated food losses of plant origin are often treated as a potential risk. Although this legislation was originally implemented to protect public and animal health by preventing the spread of diseases, it also creates regulatory barriers that limit safe recovery pathways and circular use of food resources (Rao et al. 2021), even when the actual risk is minimal or manageable. Taken together, these ambitions and their enclosed constraints underline the need for diversified and complementary recovery pathways that can collectively manage the range of surplus and by-products of varying quantity and quality arising across the food supply chain. Ultimately, aiming towards each food resource being recovered and directed to its most appropriate and sustainable use (Teigiserova et al. 2020).

3.4 Recovery pathways for surplus food and by-products

To address the challenges related to inefficient food systems, many innovative approaches are being explored to circulate underutilized resources to the food supply chain (Schieber 2017; Moreno-González & Ottens 2021). Different strategies and pathways, preferably adapted to the local conditions, are required to tackle the vast range of surplus and by-products arising along the food supply chain (Dhiman et al. 2025; Wibisono et al. 2025). Some examples, summarized in Table 2, include technology interventions to optimize production and consumption pathways (Agya 2025; Juszczak-Szelągowska et al. 2025), genetic engineering for enabling high-value valorisation of surplus (Hemalatha et al. 2023), upcycling of by-products into food or feed ingredients (Punia Bangar et al. 2024; Thorsen et al. 2025), and bioconversion using microorganisms (Kaur et al. 2023).

Table 2. Description of processing terms covered in this thesis.

Term	Description
Bioconversion	Refers to the use of biological processes, typically involving microorganisms, fungi, or insects, to transform food resources into valuable products such as food ingredients, animal feed, biofuels, or fertilizers.
Genetic engineering	Involves the direct modification of an organism's DNA to enhance specific traits or introduce new functions, such as improving the nutritional profile, yield, or environmental performance of crops.
Genome editing	A precise form of genetic engineering, often using tools like CRISPR-Cas9, to achieve desired traits without introducing foreign DNA.
Process technology	Encompasses mechanical, thermal, and chemical methods for converting raw materials or food resources into stable, safe, and functional ingredients or products within the food, feed, and bio-based sectors.
Technology interventions	Involves the application of technological solutions, such as digital tools, data-driven innovations, co-creation, market-driven actions, and mechanical interventions, to support the prevention, reuse, and recovery of food resources.
Valorisation	Refers to converting surplus, by-products, or waste into useful resources, such as food, feed, or energy.
Upcycling	Refers to a specific type of valorisation, where surplus, by-products, or other food resources are transformed into higher-value products, particularly for human consumption.

By utilizing multiple technologies and approaches for recovery, a broader range of resources can be made available for food or feed applications (Agya 2025). Such diversification could, in turn, support global sustainability objectives by contributing to more resource-efficient, resilient, and circular food systems (Lorenz et al. 2025). However, since recovery pathways might differ with respect to, e.g., resource efficiency, environmental performance, technological maturity, or applicability, it is important to assess when, and under what conditions, these pathways can support sustainable food systems.

3.5 Trade-offs with circular food systems

While circular food systems can offer many promising solutions to reduce food waste and enhance food system sustainability, their implementation is also linked to potential trade-offs and practical constraints (Vågsholm et al. 2020). Recovery pathways generally involve additional energy use, water consumption, or material inputs due to additional processing, preservation, or transport requirements (Swaraj et al. 2025; Wibisono et al. 2025). If not sufficiently balanced, these added steps could overshadow the net environmental benefits of recovering untapped food resources. Geographic and contextual factors further complicate implementation (Lotfian Delouyi et al. 2023), as not all pathways are feasible under the same conditions. Moreover, food safety and hygiene regulations can restrict the reuse of surplus or by-products (Rao et al. 2021), especially when reintroducing them in human nutrition. As also voiced by Lalander & Vinnerås (2022), while critical for protecting public health, such regulations may unintentionally prevent the valorisation and recovery of safe, edible resources.

Economic trade-offs also emerge as the cost of recovery, sorting, and reprocessing can exceed the value of the end products or reduce profit margins (Read & Muth 2021), making circular strategies less attractive without policy incentives or established market demand. These insights also echo findings by Isaac-Bamgboye et al. (2025), who further addressed the importance of considering social aspects, as potential environmental benefits may be offset by low social acceptance. In practice, this could translate into novel food products being rejected by consumers or industry actors due to perceived safety concerns or higher costs.

Another important consideration is the trade-offs between resource use. In Sweden, for instance, a considerable share of source-separated food waste is directed towards biogas production via anaerobic digestion, which in turn powers public transport (Johansson 2021). This practice was originally developed as a sustainable solution to divert food waste from landfills, by instead recovering nutrients and producing biofuels (Abrahamsson 2023). This pathway initially resonated well with the European Waste Hierarchy, as energy recovery should be favored over disposal. However, over time, it has inadvertently led to a systemic dependence on the continued generation of wasted food to sustain renewable energy production.

As modern food supply chains continue to evolve and higher-value recovery pathways gain more scientific and industrial relevance, a battle for biomass could be around the corner. A similar concern was stressed by Muscat et al.

(2020), also underlining that if more food resources are recovered for food or feed, less biomass remains available for energy. They further suggest following similar levels of priority as used in the food waste hierarchy, where available food resources are firstly considered for pathways towards food, followed by feed and bioenergy. In turn, this could reveal a new challenge: balancing competing circularity pathways and conflicting interests. The importance of considering different sustainability perspectives, covering all three sustainability aspects, was also voiced by Goldáraz-Salamero et al. (2025). Acknowledging these trade-offs is important to avoid burden shifting and navigate the benefits and limitations with different recovery pathways.

4. Materials and Methods

The thesis explores four strategies for recovering untapped food resources, spanning the three highest priority levels of the food hierarchy. All studies (*Papers I–IV*) employ LCA to quantify environmental impact. This is a widely recognized analytical framework used to evaluate the environmental impacts associated with all stages of a product's life, from the extraction of raw materials and manufacturing processes to its use phase and eventual disposal (EPLCA 2025). As a method, it gained international significance via the establishment of the ISO 14040 (2006) and ISO 14044 (2006) standards by the International Organization for Standardization. These guidelines formalized the structure and terminology of LCA, laying the foundation for consistent and transparent environmental assessments. An LCA is typically structured according to the standardized framework, illustrated in Figure 5, consisting of four iterative phases.

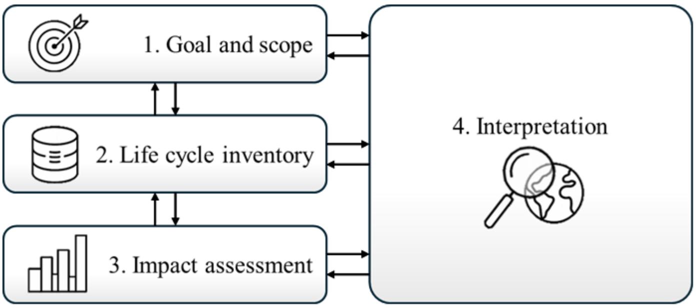

Figure 5. Illustration of the life cycle assessment framework.

The goal and scope definition establishes the aim of the study, system boundaries, and functional unit. This is followed by the life cycle inventory phase, which involves collecting data on material and energy inputs, emissions, and waste flows. Next, the life cycle impact assessment translates these flows into environmental impacts using LCA methods. Finally, the interpretation phase evaluates the findings, addresses uncertainties, and draws conclusions to support decision-making. The following section is structured using this framework, alongside complementary approaches used.

4.1 Goal and scope

The first step of the LCA framework is to define the goal and scope of the study. This stage ensures methodological consistency and alignment with the intended objectives. In this thesis, the common aim of *Papers I–IV* is to assess the environmental impacts of transitioning from conventional food management practices toward more circular pathways, via resource recovery of surplus and by-products. The scope of each study was set by the identification of the system boundary, the *functional unit* (FU), and key assumptions. A *cradle-to-gate* perspective was applied for all papers in this thesis, encompassing relevant processes from raw material acquisition, transportation, energy inputs, and processing needs, while also accounting for potential losses and waste along the supply chain. All studies adopt a physical, mass-based FU to reflect the primary function or nutritional output of each product system. This consistent approach enables comparability across different contexts while ensuring that the environmental impacts are evaluated using a purposeful and measurable quantity.

4.1.1 Methodological approach

When assessing the environmental impacts of complex systems, applying either an *attributional* (ALCA) or *consequential* (CLCA) approach can become methodologically challenging. The food systems are inherently complex, involving multiple actors, dynamic flows, and multifunctionality. ALCA typically provides a static snapshot of environmental burdens associated with a specific product or process, while CLCA aims to capture the broader system-level consequences of a change, such as introducing a recovery pathway. In practice, the boundaries between these approaches can blur when dealing with surplus food and by-products, where the environmental burden is often shared or indirect.

4.1.2 Managing multifunctionality

Food systems often produce multiple interconnected outputs, such as surplus or by-products, making the management of multifunctionality a central methodological challenge. In life cycle assessment, two main approaches are commonly used to address this: allocation and system expansion (Leinonen et al. 2025). The allocation approach manages multifunctionality by determining how the environmental burdens should be distributed among multiple outputs from a system, for instance, based on physical mass or economic value (Dominguez Aldama et al. 2023). System expansion is another multifunctionality approach in LCA. Instead of allocating environmental burdens among multiple outputs, it expands the system boundaries to include the *avoided impacts* achieved when recovered resources substitute conventional ones (Leinonen et al. 2025). When recovered food resources follow pathways towards food, feed, or bioenergy applications, they can displace the need for virgin resources. System expansion can thereby quantify the potential environmental benefits of food recovery pathways by capturing their specific substitution effect.

4.2 Life cycle inventory

A joint feature of the thesis is the inclusion of transparent *life cycle inventory* (LCI) tables describing inputs and outputs from the modelled systems. The inventory data was generated through primary data, e.g., via stakeholder collaborations, and secondary data, including previous research. Primary data was often gathered directly from industry actors and research partners, through interviews or production records, providing detailed information on raw material inputs, energy use, transport, waste flows, and by-products. The collaborations enabled robust life cycle inventories and the creation of actionable, context-aware scenarios. When primary data was unavailable, high-quality secondary data was used to complement the inventories, using previous research, national statistics, and company reports to describe the inputs and outputs needed in each system.

4.2.1 Databases

In this thesis, the standardized LCI databases Ecoinvent (versions 3.5–3.10) and Agri-footprint (versions 6.0–7.0) were employed to describe the impact of each input and output. While Ecoinvent offers broad coverage of data across multiple industrial and service sectors (Weidema et al. 2013), Agri-footprint complements this by providing high-resolution, sector-specific data for crop production, animal products, and food processing (Agri-footprint 2020), making their combined use particularly valuable for comprehensive assessments in food systems.

4.2.2 Customised datasets

Although standardized LCI databases provide a robust foundation for environmental modelling, it does not provide full coverage (Corrado et al. 2018). Customized datasets can allow improved accuracy and relevance by tailoring data to specific technologies, regions, or system boundaries. Therefore, multiple custom datasets were designed throughout the work when needed to enhance representability or improve coverage of available datasets. These ranged from accounting for increased use of fossil-free electricity and heat (*Paper I*), country-specific production of food such as egg and wheat flour (*Papers II–III*), alongside process-specific burdens of feed ingredients (*Paper IV*). This approach allowed for greater flexibility and precision in modelling of specific processes or products. By adapting inventory data, such as parameters not directly controlled by the user, to better reflect local conditions, emerging technologies, and case-specific factors, the results became more representative and decision-relevant.

4.3 Impact assessment

The third phase, *life cycle impact assessment* (LCIA), involves assessing the environmental impacts of inputs and outputs identified in the inventory phase. This procedure is done by classifying and characterizing the inventory flows into impact categories based on their contribution to specific environmental concerns (Huijbregts et al. 2016), such as climate change, eutrophication, acidification, or land use change. In this study, the LCIA was conducted using the ReCiPe or environmental footprint (EF) method, both providing a consistent and scientifically grounded framework for impact assessment, including all available midpoint indicators to gain a holistic result of the environmental impacts. Normalization and weighting steps were not applied at the midpoint level, in line with ISO 14044 (2006) recommendations for comparative studies. However, to complement the midpoint indicators and provide a measure of biodiversity impact, the weighted indicator *Ecosystem damage* was included at the endpoint level using the ReCiPe (H) method (*Papers I–IV*). This approach provided a broader perspective, without jeopardizing transparency at the midpoint level.

4.3.1 SimaPro software

All input and output data, including customized datasets, were modelled using SimaPro (versions 9.0–9.5). The software is linked to multiple databases, e.g., Ecoinvent and Agri-footprint, and provides standardized calculation setups for multiple assessment methods, such as ReCiPe and EF (PRé Sustainability 2025). Over the study period, successive software updates introduced expanded database coverage, updated emission factors, and refined methodological options, improving consistency with current environmental and impact metrics.

4.3.2 Method development

Assessing biodiversity impacts is a critical yet still underdeveloped aspect of LCA, limiting the ability to capture a key dimension of environmental sustainability (Crenna et al. 2020). Current LCA methods only capture three of the five direct drivers of biodiversity loss: land use change, climate change, and pollution, while lacking robust coverage in overexploitation and invasive species. Aquatic ecosystems remain particularly underrepresented, with ongoing method development not yet reaching standardization (Winter et al. 2017; Bromwich et al. 2025). A similar discussion was held by Damiani et al. (2023), who further emphasize the need to expand biodiversity impact assessment by improving driver coverage, integrating ecosystem services, and developing more representative indicators. This was later echoed in the findings by Bergman et al. (2025), stressing the importance of developing methods that capture overexploitation.

This methodological gap is particularly relevant for aquatic value chains, such as fish oil and fish meal production (Stanford-Clark et al. 2024; Deville et al. 2025), where overfishing and biomass depletion are well-documented environmental concerns that lack coverage in standardized LCA (Avadí et al. 2020). Given the aquaculture focus of *Papers I* and *IV,* this knowledge gap was addressed using the species-specific characterization factors by Langlois et al. (2015). This approach was applied via sensitivity scenarios to explore how overexploitation, in terms of biomass removal from aquatic ecosystems, influenced the environmental impacts.

4.4 Interpretation

The interpretation phase is the final step of an LCA, in which the results from the LCI and LCIA are systematically evaluated in relation to the defined goal and scope (Laurent et al. 2020). This phase involves identifying impacts, assessing data quality and uncertainty, and drawing conclusions or recommendations based on the findings. In this work, interpretation was used to highlight key environmental hotspots, compare alternative scenarios, and assess the robustness of the conclusions through sensitivity analysis.

4.4.1 Scenario analysis

Scenario-based modelling can bridge potential knowledge gaps between theory and practice (Sica et al. 2024; Gupta et al. 2025), enabling evaluation of how current actions or inactions might lead to different outcomes. By exploring possible pathways for recovering food resources in the European context, potential limitations, benefits, and trade-offs can be revealed and thereby sufficiently addressed before real-world implementation (Reilly & Willenbockel 2010). Using scenario analysis was motivated in this thesis for three key reasons. Firstly, business as usual is no longer an option, and we need to evaluate what available pathways could, and should, be implemented moving forward. For this, comparing alternative strategies to the current practice can be very useful, although empirical data might be scarce (Franco & Cicatiello 2021; Sukuman et al. 2025). Secondly, scenario analysis provides a way to test theoretical changes before implementing them in practice, helping to identify potential environmental impacts and risks. Thirdly, there is no joint consensus among actors, stakeholders, and policymakers operating in the Swedish and European food systems on what challenges are most urgent, and which pathways should be prioritized. Developing plausible scenarios and simulating their potential outcomes thereby allows exploration of conflicting perspectives.

4.4.2 Sensitivity analysis

In LCA, sensitivity analysis can take several forms depending on the type of uncertainty or assumption being tested. Its general purpose is to evaluate the robustness of the results and to determine how changes in key parameters or assumptions might influence the outcomes (Corrado et al. 2017). In this thesis, four main types of sensitivity analysis were performed: *parameter sensitivity*, *data sensitivity*, *scenario sensitivity,* and *methods sensitivity*. Parameter sensitivity examines how variations in quantitative input data, such as energy use or crop yield, affect the results (De Marco et al. 2018). This approach helps identify which parameters have the greatest influence on the environmental impacts and therefore might require particular attention in data collection or modelling approach. Data sensitivity evaluates how the quality, availability, or representativeness of data influences the results, and can capture uncertainties stemming from variability, limited precision, or generalization in LCI data. In practice, this could refer to testing the use of locally specific data versus default database values, or the substitution effect. Scenario sensitivity explores how different assumptions, technologies, or management pathways influence outcomes by comparing alternative system configurations. Finally, sensitivity related to methodological coverage or approaches to manage multifunctionality has been found to highly influence the environmental impacts of food systems (Djekic et al. 2019).

4.5 Quantification and collaboration

4.5.1 Mass balance and material flow analysis

Although closely related and often overlapping in practice, the material flow analysis and mass balance concepts serve slightly different purposes and operate at different levels. *Material flow analysis* (MFA) is a tool developed for mapping the movement and transformation of materials or resources through a system (Baars et al. 2022). Quantifying flows and stocks of food resources is essential to identify where inefficiencies occur and to evaluate their potential to replace conventional inputs in food systems (Barkhausen et al. 2023). In this thesis, MFA was used to identify the quantity and origin of available by-products (*Paper II*) and surplus (*Paper III*) arising along the supply chains. While MFA provides a systems-level perspective, mass balance generally provides more detailed insights at the process level. The concept of *mass balance* (MB) focuses on quantifying the inputs, outputs, and internal flows within a specific process or production stage (Dufossé et al. 2024), such as algae cultivation (*Paper I*) or production of aquafeed ingredients (*Paper IV*). By aligning input materials with output products and losses, MB can also support in generating the inventory data and support approaches for managing multifunctionality (Scherhaufer et al. 2020).

4.5.2 Stakeholders and industry actors

Collaboration among stakeholders and industry actors, and with researchers and policy makers, is considered a key lever in advancing sustainable food systems (Filimonau & Ermolaev 2021; O'Donnell et al. 2025), as it can improve the representativeness of real-world conditions and operational constraints. This was also voiced by Watabe & Takano (2025), further highlighting that while individual initiatives are at risk of limited impact, their collective impact is vital to shape sustainable future food systems. In turn, this could foster shared understandings of opportunities and challenges, making proposed pathways more credible and actionable. In *Paper I*, collaboration was concentrated to research-focused stakeholders, while in *Paper II* the collaboration was expanded to researchers developing genome editing in potato cultivars and Swedish industry actors. For *Paper III*, the collaboration was further broadened through engagement with multiple actors across the Swedish bakery sector, ranging from industrial bakeries and retailers to distribution companies and food aid organisations, alongside scientific partners from Europe. Finally, in *Paper IV*, collaboration included multiple research partners developing novel aquafeed ingredients, alongside industry actors operating within European aquaculture.

5. Results

While considerable amounts of food resources could be recovered across the studied systems, the results clearly indicate that the environmental benefits of recovery depend strongly on the pathways through which these resources are utilized. A central conclusion is that food resource recovery in general results in lower environmental impacts than their conventional counterparts, particularly with respect to climate impact, terrestrial acidification, and land use. This approach not only enhances resource efficiency but also aligns with the food waste hierarchy framework by keeping food-derived resources at a higher functional or nutritional value within the system. The results section presents the main environmental outcomes of *Papers I–IV*, evaluating how different recovery pathways influence the environmental performance.

5.1 Environmental impacts of recovered food resources

By jointly assessing the recovery pathways explored in *Papers I–IV*, this section illustrates how the choice of recovery pathway and end use influence the overall environmental performance of recovered food resources. Replacing fish oil with algae oil cultivated using wasted food as substrate (*Paper I*) yielded notable environmental benefits, especially with respect to climate impact and terrestrial acidification, which were maintained when incorporated into aquafeed (*Paper IV*). Recovery of potato protein through genetic engineering also showed substantial reductions in global warming potential, acidification, land use, and ecosystem damage, with the greatest benefits achieved when the recovered protein substituted eggs (food) rather than soybean meal (feed). Nevertheless, as demonstrated in later work (*Potato scenario in Paper IV*), considerable environmental gains could be realised through direct reuse in aquafeed. On the contrary, the results also revealed that direct reuse of bakery surplus to substitute feed did not exceed the environmental burdens of producing this resource (*Paper III*). This was also evident when incorporated in aquafeed formulations (*Bread scenario* in *Paper IV*). Instead, bakery surplus yielded the highest environmental benefits when directed toward prevention and valorisation for human consumption. However, surplus bread was found to hold a greater substitution potential when upcycled through bioconversion routes (e.g., *Algae* and *Insect scenarios* in *Paper IV*). This emphasizes that when prevention or reuse for human nutrition is not feasible, the preferred pathway for surplus bread and bakery products could be as a substrate for cultivating microalgae or rearing insects, enabling this carbohydrate resource to be converted into fat or protein biomass with a higher substitution potential.

To complement the results in *Papers I–IV*, the environmental impact of each recovery pathway (algae oil, potato protein, and surplus bakery goods) was assessed from a product perspective. In the modelling, wasted food resources were treated as burden-free, reflecting their classification as waste, whereas surplus food and by-products were assigned a fraction of the upstream environmental burdens associated with their production. In contrast, the pathway impact was calculated using equation (1) to capture environmental impacts of substituting conventional food or feed ingredients using these recovered resources. Both the product and pathway impacts are shown in Table 3, while additional impact categories are available in the *Appendix.*

$$\textit{Pathway impact} = \textit{Product impact} - \textit{Substituted product impact} \quad (1)$$

Table 3. Environmental impacts of each recoverable product (**P**), alongside net environmental impacts of using this resource to substitute (food) or (feed). Negative (green) values indicate a substitution effect larger than the impact of recovering the resource.

	Paper I / kg DHA		Paper II / kg protein			Paper III / kg carbohydrate			Paper IV / kg aquafeed			
	Algae oil (**P**)	Fish oil (feed)	Potato protein (**P**)	Egg (food)	Fish meal (feed)	Surplus bread (**P**)	Bread, prevention (food)	Animal nutrition (feed)	Algae scenario (feed)	Potato scenario (feed)	Bread scenario (feed)	Multi-source scenario (feed)
GWP	7.5×10^{0}	-2.2×10^{1}	2.5×10^{0}	-3.2×10^{0}	4.2×10^{-1}	2.9×10^{0}	-2.8×10^{0}	2.2×10^{0}	-3.6×10^{-1}	-3.2×10^{-1}	-6.0×10^{-2}	-2.1×10^{0}
TA	2.1×10^{-2}	-1.4×10^{-1}	1.1×10^{-2}	-8.7×10^{-2}	-3.6×10^{-3}	2.5×10^{-2}	-2.4×10^{-2}	1.9×10^{-2}	-4.1×10^{-3}	-1.4×10^{-3}	1.1×10^{-3}	-5.2×10^{-3}
F-EU	2.6×10^{-3}	-1.4×10^{-1}	8.2×10^{-4}	-1.6×10^{-4}	6.2×10^{5}	1.5×10^{-3}	-1.5×10^{-3}	1.2×10^{-3}	-8.9×10^{-5}	-4.9×10^{-5}	-8.3×10^{-5}	-3.6×10^{-4}
LU	2.1×10^{0}	7.3×10^{-1}	3.4×10^{-1}	-7.6×10^{0}	2.4×10^{-1}	8.2×10^{0}	-8.2×10^{0}	6.8×10^{0}	-7.4×10^{-1}	-2.6×10^{-1}	3.8×10^{-1}	-2.3×10^{0}
EQ	5.3×10^{-8}	-1.0×10^{-7}	1.9×10^{-8}	-9.3×10^{-8}	7.3×10^{-9}	9.2×10^{-8}	-9.1×10^{-8}	7.5×10^{-8}	-8.6×10^{-9}	-3.4×10^{-9}	2.7×10^{-9}	-2.9×10^{-8}

Midpoint level: GWP – Global warming [$kgCO_2eq$], TA – Terrestrial acidification [$kgSO_2eq$], F-EU – Freshwater eutrophication [kgPeq], LU – Land use [m^2a crop eq]. Endpoint level: EQ – Ecosystem Quality [species×year]

The results in Table 3 illustrate the magnitude of potential environmental savings from advancing food resource recovery across the three case studies in *Papers I–III,* and how using these resources to substitute commercial aquafeed ingredients (*Paper IV*) influences environmental impacts. Key findings show that the environmental performance was determined less by the processing and technological means required for a specific recovery pathway, such as bioconversion or genetic engineering, than by the substitution potential enabled via these pathways. This is consistent with the overarching principle of utilizing recovered resources at their highest possible value within the food system, ensuring that each resource contributes optimally to circularity and sustainability. In this sense, the results partly support the food waste hierarchy framework, where prevention and high-value reuse are prioritized; however, the results also demonstrate that lower-tier pathways, such as bioconversion for subsequent reuse in aquafeed, can be more suitable for converting carbohydrates into fat or protein. The results in Table 3 show that, with respect to bakery surplus, direct reuse in animal feed was less favourable than transforming this carbohydrate-rich resource into fat or protein via bioconversion. Recovery pathways should thus be prioritised also with respect to their ability to generate nutrients and products with higher functional or nutritional value.

5.2 Synthesis of recovery pathways for food resources

While the results for product and pathway impacts enable a more detailed analysis, a combined synthesis further allows evaluation based on their relative impacts and environmental performance in a larger context. For cross-comparison and synthesis of the findings, the *Multi-source scenario* was used as a reference point for the results in *Paper IV*. This nutritionally balanced aquafeed formulation incorporates recovered macronutrients from algae oil (*Paper I*), potato protein (*Paper II*), and bakery surplus as the main substrate for algae cultivation and insect rearing (*Paper III*).

To provide a consistent basis for comparison, the synthesis first accounts for the type, quantity, and composition of each recoverable food resource (shown in Table 4). The functional output integrates the functional product equivalent of each recovered resource, enabling comparison of results across pathways with different product functions. The amount of recoverable resource was calculated based on the quantity of product generated from the available resource. The functional output from each of the assessed recovery pathways was calculated using equation (2). Algae bioconversion yields 1% recoverable resource, while 100% of the available potato protein can be recovered via genome editing. For surplus bread, 22% of the available resources were directed to prevention and food donation, while the remaining quantities were the recoverable resources.

$$Functional\ output = Recovered\ resource \times Macronutrient\ content \qquad (2)$$

To complement the environmental impacts assessed per kg in Table 3, the environmental performance was calculated by scaling these impacts to a national perspective using equation (3). Four midpoint impact categories and one endpoint damage category were considered in the environmental performance. Thereby, the available amount of resource, its nutritional or functional properties, alongside substitution effects when incorporated as a food or feed resource, was considered as a complement to product impact.

$$Environmental\,performance = Pathway\,impact \times Functional\,output \quad (3)$$

While environmental performance captures the potential benefits of each pathway, it does not reflect how close these solutions are to practical implementation. Therefore, the technological readiness level (TRL) was included as a complementary indicator to assess the maturity and feasibility of each recovery approach. Following the European Commission et al. (2025) framework, the TRL for each recovery pathway was assessed from level 1 (basic research) to level 9 (commercial application). Table 4 summarises the available and recoverable resources, their functional outputs, environmental impacts, and technological readiness for each pathway.

Table 4. Synthesised results of thesis papers.

Description	**Paper I**	**Paper II**	**Paper III**	**Paper IV**
Type of food resource	Food losses	Potato starch by-product	Surplus bread and bakery goods	Recovered surplus and by-products
Available resource [ton/year]	643 000	1 500	90 000	734 500
Recovery pathway	Bioconversion	Genetic engineering	Technology interventions	All (*Papers I–III*)
Main macronutrient	Fat	Protein	Carbohydrates	All
Main product via recovery	Algae oil	Potato protein	Surplus bread	Novel ingredients
Recovered resource [ton/year]	6 700	1 500	70 000	78 200
Macronutrient content	40%	80%	50%	47%
Functional unit	1 kg omega-3	1 kg protein	1 kg carbohydrate	1 kg aquafeed
Functional output [ton]	2 680	1 200	32 900	36 780
Food waste hierarchy level	3	2	1 – 3	3
Environmental performance				
Carbon footprint [tonCO_2eq]	-5.9×10^{4}	-3.8×10^{3}	6.1×10^{3}	-7.8×10^{4}
Terrestrial acidification [tonSO_2eq]	-3.8×10^{2}	-1.0×10^{2}	5.1×10^{2}	-1.9×10^{2}
Freshwater eutrophication [tonPeq]	-2.1×10^{1}	-2.0×10^{-1}	7.8×10^{0}	-1.3×10^{1}
Land use [m^2a crop eq]	2.0×10^{3}	2.9×10^{2}	-5.8×10^{4}	-8.4×10^{4}
Ecosystem quality [species × year]	-2.8×10^{-4}	8.8×10^{-6}	-4.0×10^{-4}	-1.1×10^{-3}
Technological readiness level	4 – 6	2 – 4	8 – 9	6 – 8

The results reveal notable differences in scale, environmental performance, and technological maturity of the assessed recovery pathways. Bakery surplus (*Paper III*) reached the highest TRL, reflecting well-established industrial practices and logistics for surplus management in Sweden. This pathway inferred considerable environmental benefits via prevention and donation of surplus; however, the environmental burdens of production and reprocessing to make surplus bread available for feed substitution were greater than the benefits achieved via substituting feed ingredients. This explains the positive values for climate impact, terrestrial acidification, and freshwater eutrophication observed in Table 4. Among macronutrients, fat and protein generally have higher environmental impacts due to the resource intensity of their production and processing (Poore & Nemecek, 2018), and the notable environmental benefits of substituting these resources was therefore expected.

A similar trade-off was observed for land use change when upcycling potato protein (*Paper II*), which otherwise achieved considerable environmental benefits, since the substitution of conventional food products outweighs the impacts of recovery. Moreover, genetic engineering currently holds the lowest TRL (2–4) among the assessed pathways, reflecting its early-stage development and limited large-scale implementation compared to more established recovery pathways. Reuse of recovered food resources in novel aquafeeds (*Paper IV*) demonstrated a relatively high TRL (6–8), supported by existing pilot operations integrating multiple recovered ingredients. Its strong environmental performance, particularly in terms of climate impact and land use, indicates considerable untapped environmental benefits. Algal bioconversion (*Paper I*) showed moderate TRL (4–6) and considerable trade-offs in environmental impacts, with substantial climate benefits but increased impact on land use. Despite the large availability of wasted food and food losses, only a small fraction can be converted into algae oil, limiting its overall resource recovery efficiency. On the contrary, while genetic engineering exhibited the lowest TRL and small quantities of available potato starch by-products, this pathway enabled the highest nutrient yield.

Among the evaluated pathways, reuse in aquafeed demonstrated the highest environmental performance on the national level, likely driven by two main factors: the favourable substitution of fat and protein leading to substantial environmental credits, and the mitigation of pathway-specific trade-offs through the integration of multiple, complementary recovery routes. At the same time, the results in Table 3 clearly show that prevention and high-value valorisation are preferable from an environmental point of view, especially given that the only way to reduce environmental impact is by avoiding it to begin with. Achieving zero food waste remains an important long-term goal; however, it cannot be considered practically attainable within the near future, as there is an upper limit to how much food resources can be prevented and donated. These constraints are influenced by a range of factors, such as market imbalances, consumer demands, and unavoidable losses inherent to global food supply chains. In turn, this points to the need for alternative recovery pathways that enable reuse at the highest potential value, also after prevention and donation pathways are exhausted.

These findings bring forward one of the key messages from the synthesis: that no single recovery pathway can optimise both environmental and technological performance for all different food resources. Instead, the combined utilisation of complementary recovery pathways presents the most preferable route toward advancing resource-efficient food systems.

While technology interventions focused on prevention and donation, exemplified via bakery surplus to human nutrition, can deliver measurable environmental gains in commercial applications, emerging bioconversion and genetic engineering pathways offer complementary routes for valorising and upcycling otherwise wasted food resources, provided that technological maturity is enabled. These insights point to a clear need for both targeted development and implementation of recovery pathways to bridge the gap between environmental potential and practical feasibility, thereby unlocking the inherent environmental benefits of advancing food resource efficiency.

5.3 Unlocking the untapped potential in food resources

The transition toward more circular and resilient food systems requires not only reducing food waste but also the implementation of efficient recovery pathways that allow resources embedded in the food supply chain to be utilized at their highest potential value. Large volumes of surplus and by-products are currently downcycled, diverted to energy recovery, or discarded without valuable recovery, despite their potential to replace imported and environmentally intensive food and feed. Understanding the quantitative potential of these resources is crucial to guide policy and investment toward effective strategies and to identify where, and under what conditions, recovery can support food security and environmental targets. Developed from the combined findings of *Papers I–IV* and the overarching synthesis of this thesis, Figure 6 illustrates a future outlook on implemented recovery pathways on a national level, using Sweden as an example.

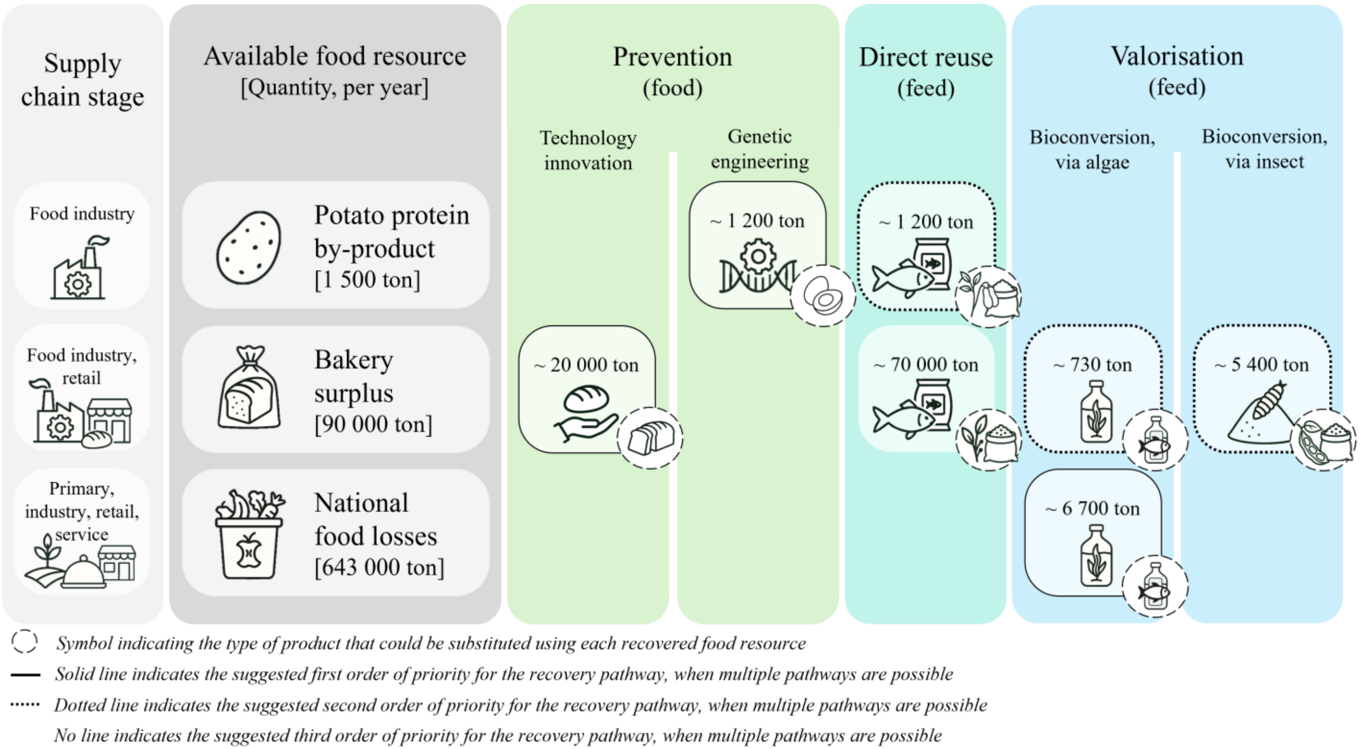

Figure 6. Overview of recoverable food resources in Sweden, illustrating annual quantities and opportunities for reintegration into food and feed production through recovery pathways following the three highest priority levels in the food waste hierarchy.

By suggesting levels of priority, from an environmental perspective, the figure provides an outline for how to navigate between multiple pathways to enable recovery in accordance with the highest potential value of each assessed resource.

Roughly 1 000 tonnes of fish oil is needed to sustain Swedish rainbow trout cultivation annually, given a 15% inclusion rate in commercial aquafeed (Langeland et al. 2023). Using the fatty acid content as a substitution baseline, roughly 100 000 tonnes of wasted food resources directed towards algae oil production would, in theory, be sufficient to replace 100% of the fish oil required. In turn, this could considerably reduce dependence on imported marine oils and substantially mitigate ecosystem pressures associated with aquafeed (Majluf et al. 2024). A national supply of algae oil combined with anaerobic digestion, as illustrated in *Paper I*, would not create competition between feed and energy, since both can be produced from the same quantity of wasted food resources. Although full replacement of fish oil in aquafeed might not yet be feasible, as suggested in *Paper IV*, the untapped potential of producing algae oil from wasted food is clear.

In Sweden, approximately 404 000 tonnes of potato starch are produced annually in Sweden (Swedish Board of Agriculture 2024), generating over 1 500 tonnes of protein by-products every year. Using genetic engineering pathways, such as the CRISPR-Cas9 approach, this resource could substitute egg protein in foods (*Paper II*) or, without editing, replace roughly 1 700 tonnes of protein in aquafeed (*Paper IV*). One of the benefits of this pathway is that it allows industry processes to remain largely unchanged, which could be particularly attractive from a market perspective, as it creates value without major infrastructural adjustments (Kalaitzandonakes et al. 2023). Comparable protein yields could also be achieved by optimising industry processes, e.g., reducing heat intensity and lowering glycoalkaloid content.

At the food industry and retail levels, roughly 90 000 tonnes of surplus bakery products are generated annually in Sweden, where only 20% could be avoided via prevention measures and 2% reused via food donations (*Best Practice scenario* in *Paper III*). The remaining 70 000 tonnes available for feed recovery held the highest environmental benefits when converted into fat or protein via algae or insects (*Paper IV*). In theory, this quantity of surplus could yield up to 730 tonnes of algae oil (calculated from *Paper I*) or over 5 000 tonnes of defatted insect meal (calculated from *Paper IV*). Such recovery potential corresponds to the theoretical replacement of up to 70% of fish oil or over 100% of the soybean protein required to sustain Swedish rainbow trout production. Together, these case studies show that Sweden alone holds a recovery potential exceeding 36 000 tonnes of fat, protein, and carbohydrate annually (Figure 6). These results further suggest that when prevention and direct reuse as food are not feasible, pathways aimed at producing fat and protein could be prioritized to maximize the environmental benefits of resource recovery. If effectively recovered and reintegrated, these food resources also could play a strategic role in advancing resource efficiency and mitigating environmental pressures across the food-feed nexus. It should be noted, however, that the case studies assessed in this thesis only represent a small fraction of the food resources and recovery pathways currently available or emerging in Sweden and Europe. Consequently, the true recovery potential is likely considerably higher, underscoring the need for future studies to expand the assessment to capture the broader range of food resources and recovery pathways.

6. Discussion

This thesis demonstrates that recovering surplus food and by-products to reintegrate them into the food supply chain can substantially reduce environmental impacts. Beyond quantifying their recoverable potential, the results highlight the practical and systemic conditions required to realize this value, including technological maturity, environmental benefits and trade-offs, market integration, and policy support. The findings collectively underscore the need to reframe these materials not as waste, but as valuable resources within sustainable food systems.

6.1 Quantifying recoverable food resources

As modern food supply chains continue to evolve, the current distinction between *food waste*, *wasted food*, and *food resources* will likely become increasingly blurred. In turn, this emphasizes the need for joint definitions that reflect both the original intent of production and the realized value of recovery. Arguably, this can be considered a prerequisite for advancing circularity and identifying the true scale and composition of recoverable resources on a national and European level. This insight can be exemplified via *Paper III*, where the scientific framework considered all products that deviated from their intended pathway as a wasted resource, meaning that also animal feed and food donations were considered a waste. This approach recognizes that such pathways do not fully retain the inherent purpose or functional value of the food resource. Consequently, from a definitional perspective, a larger share of food would be categorized as waste following this approach compared with studies or industry actors that set the boundary at anaerobic digestion or animal feed. Similar concerns have been voiced in previous research (Chaboud & Daviron 2017; Chauhan et al. 2021; Singh et al. 2021), showing that definitions of food loss and waste strongly shape both measurement outcomes and the interpretation of circularity efforts.

Moreover, the quantification in *Paper III* revealed notable discrepancies between aggregated food resource statistics and sector-specific data, as well as considerable uncertainty surrounding both the quantity of bakery surplus generated and the recovery pathways employed to manage it. If such uncertainties are evident for a staple food product like bread and baked goods, this implies that a substantial portion of resources across the wider food system likely remains unaccounted for. This lack of robust and reliable data has been reported as a major limitation in establishing quantitative targets and actionable measures for the prevention and reduction of food waste (Soloha & Dace 2025). Given that food waste quantification measures become mandatory in the EU under Delegated Decision 2019/1597, alongside legally binding food waste reduction targets under Directive (EU) 2025/1892, improving methodological consistency and data quality is essential to ensure the accuracy, comparability, and policy relevance of reported figures. These insights are echoed at both national and European levels, where such uncertainties continue to challenge the reliability of reported food waste data (Corrado et al. 2019; Baquero et al. 2023).

While national estimates suggest roughly 1.2 million tonnes of food waste annually in Sweden, of which roughly 0.64 million tonnes (54%) origin before household level (Hultén et al. 2024), sector-specific findings by Lindow et al. (2024) rather suggests 450 000 tonnes of wasted food resources spanning just eight product groups. Complementary research by Eriksson et al. (2021) further adds that 21 000 tonnes of unharvested broccoli, primarily leaves and stems, could also be recovered each year. Alongside the findings in *Papers II–III*, nearly 100 000 tonnes of surplus bakery goods and potato protein arise annually before the household level. In total, these eleven product groups amount to 0.57 million tonnes, which, compared to the national estimations, would account for 89% of the wasted food resources before the household level. Given the limited sectoral coverage of these studies and the likelihood of additional unrecorded flows, this correspondence appears unrealistic and suggests underestimations of national data. This conclusion aligns with European findings, stressing that insufficient quantification data and methodological inconsistencies could conceal substantial discrepancies both between countries and at the product level (Hartikainen et al. 2017; Hoehn et al. 2023; Soloha & Dace 2025).

Bridging this knowledge gap will thus require systematic mapping and quantification of surplus food and by-products across the value chains, with previous research voicing the importance of considering, for instance, the geographical location of food resources, their nutritional composition, safety characteristics, and seasonal fluctuations (Caldeira et al. 2019; Corrado et al. 2019; Chauhan et al. 2021). Through such comprehensive mapping, a first step can be taken from conceptual aspiration to strategic implementation of food resource recovery pathways.

6.2 Realizing the potential of surplus and by-products

A central question emerging from this thesis is: what should be done with the resources we already have? The answer to this boils down to the need for reconceptualizing these resources not as waste, but as valuable inputs within our food system (Wibeck et al. 2022; Goldáraz-Salamero et al. 2025). This echoes arguments from previous research, emphasizing that wasted food must be treated as a resource rather than a disposal problem (Teigiserova et al. 2020; van Zanten et al. 2023; Wibisono et al. 2025).

Surplus food, agricultural by-products, industrial side streams, and food losses hold a promising potential for environmental benefits, yet they are frequently underutilized or diverted into low-value applications. This represents a missed opportunity, particularly in light of growing pressures on global food systems to reduce environmental impacts while meeting rising demand for high-quality food within planetary limits. Illustrating the environmental benefits and potential trade-offs with different recovery pathways could help drive the necessary transition towards resource-efficient food systems. For example, replacing imported fish oil in aquafeed with algae oil produced from recovered food resources holds the potential to reduce climate impacts by over 30% per kg oil, while simultaneously lowering dependence on imported marine ingredients (*Paper I*). Upcycling potato protein to substitute food ingredients holds the potential to reduce climate impact by 6.7 kgCO_2eq per kg protein compared to conventional reuse in animal feed (*Paper II*). Similarly, diverting surplus bread and bakery products from energy to prevention, reuse as food or feed, holds the potential to reduce climate impact by 1.0, 0.5, and 0.3 kgCO_2eq per kg, respectively (*Paper III*). These examples illustrate how targeted recovery can create tangible environmental benefits across multiple sectors, which is supportive of previous recovery research (Lehn & Schmidt 2023; Siddique et al. 2024).

Technological development and implementation have consistently changed how wasted food resources are valued (Wibisono et al. 2025), and will likely continue to shape the future food systems. Pathways such as bioconversion and genetic engineering have already enabled recovery into a wide range of novel products, such as functional ingredients in bakery goods (Mia et al. 2025), protein products and meat analogues (Brancoli et al. 2021; Ray et al. 2023), and fermented food (Salas-Millán & Aguayo 2024). Co-product upcycling, joint certification approaches, and consistent labelling standards are some of the solutions identified for overcoming barriers towards large-scale implementation (Swaraj et al. 2025). Together, these previous developments show that while the technical potential is evident, realising it requires alignment between innovation, regulation, and market demand, which are factors valid for many food and feed resources.

Recent advances also demonstrate that upcycling food waste for microalgae cultivation is progressing toward commercial applications, even when integrated with biogas production or other waste-management systems (Wu et al. 2025). By creating multi-output systems that yield both energy and algae oil as valuable products from the same feedstock, overall resource efficiency can be maximised. In turn, this approach aligns closely with existing biogas and food waste infrastructure in many European countries, including Sweden. Similarly, the recovery of potato protein by-products is advancing through both genetic engineering and process improvements (Hu et al. 2025; Sapakhova et al. 2025). For bakery surplus, prevention measures via technological innovations and co-creation among food industry actors show particular promise for unlocking higher-value applications beyond animal feed (Eriksson et al. 2025; Melesse & Orrù 2025). Only when prevention pathways and valorisation towards human nutrition are exhausted, advancements in bioconversion, e.g., via insects reared on bakery surplus, point to this becoming a plausible recovery pathway to complement high-value valorisation (Lehn & Schmidt 2023; De Laurentiis et al. 2024). Finally, the development of aquafeeds using upcycled ingredients, such as algae oil from wasted food and agricultural losses, potato protein, and insects reared on bakery surplus, could provide a viable outlet capable of absorbing multiple food resources of varying origin and quality (Cottrell et al. 2025).

Understanding how these principles translate into measurable environmental outcomes requires analytical tools capable of testing alternative recovery strategies, such as scenario analysis (Sica et al. 2024; Gupta et al. 2025). Scenario-based modelling has been shown to support bridging gaps between theory and practice, in turn allowing policymakers to identify the most effective interventions even when empirical data are limited (Franco & Cicatiello 2021; Sukuman et al. 2025). By modelling different recovery pathways, as demonstrated in *Papers I–IV*, this approach could be expanded to other recovery pathways to enable further comparison of environmental benefits under varying conditions. Important to note, however, is that these results should primarily be understood as exploratory rather than predictive, serving as decision support rather than forecasts. A similar conclusion was also voiced by Laurent et al. (2020).

Key aspects to translating these potentials into reality are to ensure that resource inputs are made available for recovery, that outputs are valorised at their highest possible value, and that food waste is minimized by design. Similar insights have been shared by Swedish Food Agency et al. (2025) and at the European level (FAO 2025; Rockström et al. 2025). Still, the realization of circularity in food systems remains challenged, for instance, due to regulatory and technical barriers (Gordon et al. 2022; Philippidis et al. 2025). Inconsistent raw-material quality, heterogeneous inputs, consumer perception barriers, and insufficient regulatory clarity continue to limit the scaling of resource recovery and upcycling to commercial application (Swaraj et al. 2025). Since many of these barriers resemble those reported for managing food losses (Ishangulyyev et al. 2019), such as limited technological capacity, weak supply chain coordination, and restricted market access, it is reasonable to assume that similar constraints also will affect the recovery of surplus and by-products. Ghahremani-Nahr et al. (2025) further highlight logistical aspects, including recovery-facility location, inventory management, vehicle routing, and distribution scheduling. Overcoming these obstacles will require coordinated actions across multiple sectors. This includes developing and implementing pathways for high-value recovery; strengthening collaboration among industry actors; ensuring that outputs from one process or industry can serve as inputs to another; establishing regional collection and processing hubs that connect otherwise wasted food resourced with reuse opportunities; and revising regulatory frameworks and policy recommendations.

In practice, this could translate into food industries and retailers partnering with donation centres or feed producers to co-create novel products, such as beer, breadcrumbs, granola, insect meal, and flours. Digital platforms could also be expanded to connect retail and food service with social organisations or ingredient manufacturers, ensuring that edible surplus is directed to human consumption whenever possible. When prevention or reuse as food is not possible, recovery pathways enabling the production of protein and fat should be considered due to their environmental performance. Meanwhile, regional processing hubs equipped with sorting, drying, or fermentation technologies are needed to manage seasonal flows from primary production and the food industry. These hubs could serve as important connection points, where surplus food and by-products are collected, stabilized, and redistributed towards suitable recovery pathways. Aquafeed producers could support circularity by integrating algae oil, potato protein, and insect meal reared on surplus and by-products into commercial feed formulations, advancing both circularity and feed self-sufficiency at the national level. In turn, policymakers could incentivize these transitions by introducing recovery credits or tax benefits for companies that redirect edible food to human or feed use, combined with clearer legal definitions that classify safe, recovered food materials as resources rather than waste.

Circular and resilient food systems rarely become feasible in isolation; on the contrary, they operate within complex regulatory frameworks, market systems, and social contexts that either enable or constrain their implementation. Realizing this potential requires a shift away from large and centralized, one-size-fits-all recovery pathways (such as anaerobic digestion and incineration) to multi-pathway systems. Similar arguments have been raised by Gordon et al. (2022) and Teigiserova et al. (2020), who advocate for diversified recovery pathways tailored to local resource characteristics. In turn, this will likely entail a new set of challenges, ranging from increased need for local recovery infrastructure to regulatory hurdles and uneven market incentives. This concern echoes the *circular economy rebound effects* presented by Zink & Geyer (2017), where strategies intended to enhance sustainability may introduce new inefficiencies or social inequalities. If these challenges are not adequately considered and mitigated, even well-intended recovery pathways are at risk of looking good on paper but being impractical, unfair, or even counterproductive in practice.

6.3 Environmental, Social, and Economic Trade-offs

Although the environmental advantages of food resource recovery are well established in the present work, achieving its full sustainability potential calls for broader consideration. As highlighted by Thorsen et al. (2024) and Lehn & Schmidt (2023), assessments should preferably integrate social and economic dimensions alongside the environmental impacts. Such a holistic approach enables the identification of valorisation pathways for surplus food that optimize performance across all three pillars of sustainability.

From an environmental perspective, the results of this thesis confirm that recovering surplus and by-products generally offer clear environmental benefits by avoiding wastage of valuable resources. Similar findings have been reported by Ganesh et al. (2022) and Caldeira et al. (2019). However, environmental trade-offs remain. As shown in Table 3 and by previous research (Scherhaufer et al. 2020; Weber et al. 2023; Remijnse et al. 2025), factors such as transportation, higher energy use, and resource inefficiencies could offset potential benefits. These trade-offs are particularly relevant for perishable resources that require stabilization, freezing, or refrigeration, as also highlighted by Eriksson & Spångberg (2017). Important to consider are also methodological limitations inherent to LCA, including data quality inconsistency, uncertainties in modelling substitution effects, and the use of arbitrary system boundaries (Vadenbo et al. 2017; Domingo-Morcillo et al. 2024; EPLCA 2025). One example is the methodological uncertainty associated with the limited coverage of biodiversity impacts in LCA (Damiani et al. 2023; Bromwich et al. 2025). This was found to considerably underestimate the environmental burdens linked to food and feed resources (*Papers I and IV)*, which in turn risks distorting impact comparisons and may inadvertently misguide policy actions or industry priorities.

From an economic lens, the valorisation or upcycling of food resources can generate new value streams, reduce disposal costs, and enhance overall resource efficiency. Previous research has further demonstrated that recovery at scale is generally not limited by technological capability, but rather profitability, infrastructure gaps, and inconsistent market incentives (Babbitt et al. 2022; Dhiman et al. 2025; Yontar 2025). This emphasises another key issue, since this profitability often remains highly context dependent. The feed sector provides a striking example: although recovered food resources, such as surplus bread or potato protein, can technically be used to substitute conventional feed ingredients, factors such as strict quality requirements, limited infrastructure for small-batch processing, and higher economic investment relative to the expected return, constrain market uptake. The lack of dedicated logistics networks and the relatively low price of conventional raw materials further reduce the competitiveness of recovered resources. In *Papers I* and *IV*, this was also evident as both algae oil and insects reared on surplus food were often too expensive in comparison to already commercial feed ingredients, despite their environmental advantage. Overcoming these economic barriers will require innovative business models, collaboration between stakeholders and industry actors, alongside policy instruments that internalize environmental benefits, such as subsidies.

The social and legal dimensions of food resource recovery are equally critical moving forward. Public trust, consumer acceptance, and regulatory clarity largely determine whether surplus food and by-products can successfully be integrated into the food and feed markets. For instance, strict interpretations of food safety legislation often hinder redirection of wasted food to higher-priority levels, even in cases where scientific evidence indicates minimal risk (*Paper I*). Similarly, the use of genome-edited crops or bioconversion processes remains politically sensitive, despite their potential to enhance resource efficiency (*Papers II* and *IV*). This tension between safety and circularity has been criticized in previous work, generally emphasizing that overly precautionary regulations can hinder otherwise sustainable solutions

(Hjort et al. 2021; Girotto & Scapini 2025; Meijer et al. 2025). Previous research further suggests that the social acceptance of these innovations depends less on perceived risk and more on the degree of trust established between consumers and actors for change fostered through transparent communication (Peschel & Aschemann-Witzel 2020; Siegrist & Hartmann 2020). Giacalone & Jaeger (2023) report that consumer acceptance remains low across many recovery pathways, while Hellali et al. (2023) show that framing recovered food resources in environmental or health terms, rather than economic ones, has a stronger positive influence on acceptance.

The environmental potential of recovery strategies is clear, but their capacity to advance resource efficiency in large-scale contexts also depends on navigating these broader systemic conditions. Lotfian Delouyi et al. (2023) suggested that multi-criteria decision-making methods could serve as valuable tools for balancing these aspects, while Thorsen et al. (2025) and Moores et al. (2025) advocate the use of sustainability assessment tools such as life cycle sustainability assessment (LCSA) combined with eco-indicators to account for all three pillars of sustainability. Future food policies should therefore embed frameworks that account for environmental efficiency, economic feasibility, and social acceptability alike, ensuring that food system transitions are both just, effective, and resilient over time.

6.4 Closing the loop to strengthen the food supply chain

While much of this thesis has focused on the environmental implications of recovering surplus food and by-products, closing the loop in food systems extends beyond impact reduction alone. It also concerns building resilience, enhancing food security, and strengthening the overall functionality of the food supply chain. Beyond environmental gains, food waste prevention via resource recovery has been shown to support the resilience of food systems (Bajželj et al. 2020). This dual function, combining waste prevention with resilience building, has been further identified as a key advantage of circular food systems (FAO 2025; Rockström et al. 2025).

Achieving these benefits, however, requires a shift in both policy and public perception of what constitutes a valuable resource, especially since modern food systems tend to prioritize industrial efficiency and global trade over resilience and resource reuse. This critique echoes arguments from Gordon et al. (2022), who note that efficiency-oriented paradigms can undermine resilience and ecological sustainability. A similar concern arises when circular systems become dependent on the continued generation of wasted resources (Oroski 2025). Recovery pathways that rely on large quantities of food resources being wasted are at risk of reinforcing inefficiencies rather than solving them. To avoid this *waste-resource paradox*, recovery must be complemented by upstream prevention and dynamic downstream strategies (Teigiserova et al. 2020; Greer et al. 2021). A similar related tension is observed via the *prevention-paradox,* capturing the contradiction between widely endorsed reduction targets and the dominant industry and policy responses that prioritize waste management over prevention (Messner et al. 2020; Johansson 2021). Overcoming these challenges, linked to unlocking untapped food resources, will require valuing food for its social and environmental significance, not primarily its market price.

To guide such transitions effectively, future research should integrate social and economic dimensions alongside environmental impacts, enabling assessment of not only environmental performance but also equity, feasibility, and societal acceptance of recovery strategies. By allowing maintained nutritional or functional value, preferably in accordance with the higher levels of the food waste hierarchy, recovery pathways can play a vital role in supporting sustainable food supply while protecting the ecosystems that sustain it (Gerten et al. 2020; Richardson et al. 2023).

6.5 Key recommendations

This thesis demonstrates that food waste reduction by recovering surplus and by-products in Sweden and Europe represents tangible and evidence-based pathways for reducing the environmental impacts of our food system. Across *Papers I–IV*, the results consistently show that redirecting food resources toward higher-value uses, such as food or feed, yields substantial environmental and resource savings compared to current recovery pathways. To translate this potential into practice, three main priorities emerge:

- **Prioritise recovery pathways with high substitution effect** The main driver of environmental benefit is not recovery itself, but what the recovered resource replaces. Recovering surplus food and by-products to substitute food resources (*Papers II* and *III*) or feed ingredients (*Papers II* and *IV*) consistently reduced environmental impacts compared with pathways toward lower priority levels. This is in line with previous research and support the conclusion that food systems should be organised to retain resources at their highest functional value. This might not necessarily infer following a strict food waste hierarchy but rather entail that each resource is recovered according to its highest potential value. In practice, this could translate into source-separated resources and homogeneous materials arising at primary production or food industry level having priority towards for direct reuse or ingredient recovery, whereas mixed or lower-value food resources arising at retail or service level might achieve a higher product value through bioconversion into high-quality protein or fat. When prevention, direct reuse, or high-value valorisation is not feasible, such pathways can still deliver substantial environmental gains, provided they replace conventional inputs. While some recovery routes may currently be limited by legislation or technological readiness, this should not by default discourage their further exploration but rather encourage actions and further research to assess their potential.

- **Establish consistent definitions and expand quantification** Large discrepancies between national food waste statistics and sector-level assessments indicate that a substantial share of recoverable food resources is still undetected, and thus not made available for recovery. In turn, this can be traced back to insufficient definitions and a lack of robust quantification data. Firstly, without harmonized definitions of what constitutes as wasted food, surplus, and by-products, preferably reflecting both its intended use and actual recovery potential, it is not possible to quantify available resources. Clear, shared definitions should thus be a prerequisite, not an afterthought. Secondly, systematic quantifications capturing the magnitude of available food resources, both at a national and European level, should also consider factors specific to each resource, such as nutrient composition (protein, fat, carbohydrate), safety constraints, geographical distribution, and seasonality. Such data are essential to (i) estimate realistic substitution potential, (ii) avoid double counting across sectors, and (iii) support robust life cycle assessment.

- **Enable recovery pathways aligned with circular priorities** A fundamental aspect of moving from theory to action is determining how much of each recoverable food resource is suitable for different recovery pathways and enabling these pathways in practice. This requires dedicated recovery infrastructure, ranging from regional processing hubs, partnerships between actors, and logistics systems that keep edible surplus in the food chain. Such infrastructure should be designed to facilitate outputs from one process to become inputs in another, and so that high-value uses (food and feed) are prioritised over low-value pathways (such as energy recovery). Policy instruments should support this, e.g., via incentivising recovery pathways that displace primary production, and clarifying how recovered food resources should be managed to maintain its inherent value and avoid degradation to waste. Navigating and, ultimately, implementing these pathways should not be limited by lack of infrastructure, knowledge gaps, or insufficient economic incentives, but purposely be balanced with respect to environmental benefits, circular priorities, and market demand.

Recovery pathways should be viewed as a key component of a broader resilience agenda, especially since recovered food and feed resources can reduce import dependency and buffer against global market volatility. To fully capture the recovery potential, future scenario modelling should include not only environmental but also social and economic indicators to cover all three pillars of sustainability. Achieving this requires cross-sectoral governance that values food not merely as a commodity but as a shared, regenerative resource within planetary boundaries. In conclusion, the work presented in this thesis underscores that the environmental gains of advancing efficient resource recovery pathways are neither automatic nor uniform: they depend on how recovered resources are defined, managed, and integrated within the food system. In prioritizing high-value recovery, improving quantification coverage, expanding supportive infrastructures, and aligning policy and perception with sustainability objectives, we can move closer to realizing circular and resilient food systems where no valuable food resources are wasted.

7. Conclusion

This thesis provides a system-level understanding of how, and under what conditions, surplus food and by-products can be recovered and reintegrated into the food supply chain to reduce the environmental impacts of our food system. Across the four studies, the findings show that both the magnitude and nature of potential environmental benefits depend on enabling the most efficient recovery pathway for each food resource, accounting for their potential to substitute high-value food or feed resources.

In *Paper I*, the cultivation of microalgae using mixed food resources as substrate showed that replacing fish oil with algae oil can reduce climate impacts by roughly 75% and marine ecosystem pressures by 66%. Similarly, *Paper II* demonstrated that potato protein made available for high-value recovery through genetic engineering can substantially reduce climate impacts by 56% and ecosystem damage by 86% per kg protein when substituting food ingredients. *Paper III* revealed that prevention and reuse are the most favourable pathways for reducing environmental impacts related to surplus bread and bakery products, as these pathways reduced e.g., climate impact by over 90% compared to animal feed. Finally, *Paper IV* integrated these findings at product level, demonstrating how algae oil, potato protein, and bakery surplus can be integrated as novel ingredients in nutritionally balanced aquafeed. In turn, using these resources to substitute imported fishmeal and soybean protein resulted in notable environmental benefits per kg aquafeed compared to conventional practice, including 76% lower climate impact, 84% less land use, and 79% less damage to ecosystems.

Together, the four papers show that the environmental benefits of food resource recovery arise not primarily from recovery itself, but from reducing food waste by using surplus food and by-products to replace primary production of food and feed resources. Although recovery pathways entail additional energy and material inputs, the results demonstrate their potential to advance resource efficiency in food systems and deliver substantial environmental benefits, both in Swedish and European contexts. These findings point toward a clear opportunity: large volumes of surplus and by-products represent a considerable untapped resource within the food system. Key aspects to translating these potentials into reality are ensuring that food resources are made available for recovery, that outputs are valorised at their highest possible value, and that food waste is minimized by design.

8. Future outlook

This thesis has demonstrated the potential of circular strategies to reduce environmental impacts by recovering surplus and by-product food resources. However, several knowledge gaps remain that limit their widespread implementation and upscaling. The following section highlights research objectives to advance resource-efficient food systems and improve the environmental performance of food resources, in Sweden and beyond. Taken together, these research directions emphasize the need for an integrated and interdisciplinary approach by combining improved quantification, optimized recovery design, and broader sustainability assessment.

- **Quantifying available food resources, surplus, and by-products**
  Current data on wasted and underutilized food resources are often fragmented, outdated, or incomplete, particularly for surplus and by-products suitable for recovery toward food or feed applications. Future studies should aim to develop high-resolution inventories of these resources, both in terms of quantity and quality, across primary production, food industry, retail, and service sectors.

- **Designing and optimizing different food recovery pathways**
  Building on the findings of this thesis, future research should focus on how various food resources, such as source-separated surplus, mixed food waste, and nutrient-rich by-products, can be matched with the most suitable recovery pathways and end uses. This includes investigating the environmental and practical implications of directing resources toward alternative uses (e.g., food, feed, or energy), while ensuring alignment between recovery efficiency, product demand, and regulatory frameworks.

- **Integrating social and economic impacts into assessments**
  While this thesis demonstrated that redirecting resources from otherwise wasted food, into food or feed can yield environmental savings, such strategies may also involve trade-offs related to costs, equity, or societal acceptance. To fully capture the implications of circular interventions, future studies should integrate social and economic dimensions alongside environmental performance.

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Popular science summary

Every year, enormous amounts of edible food are lost or wasted. Globally, nearly one-third of everything produced never reaches a plate, while on a European level around 14% of all food produced is lost even before retail level. In Sweden alone, more than one million tonnes of solid food are discarded annually, much of it still suitable for consumption or high-value recovery in food or feed applications. At the same time, food production places growing pressure on the planet, contributing to climate change, biodiversity loss, and the depletion of natural resources. The challenge is clear: to feed a growing global population within planetary limits, we must make better use of the food and resources we already have.

This thesis assesses how the recovery of surplus food and by-products can support reducing environmental impacts related to food production, which in turn can strengthen the resilience of food supply chains. Instead of viewing these resources as waste, the work investigates how, and under what conditions, they can become valuable inputs for new food and feed products. Using environmental life cycle assessment (LCA), several recovery pathways were analysed: from converting food residues into new ingredients through bioconversion and genetic engineering to managing surplus production in bakeries and reusing recovered resources in aquafeed.

The results show that these recovery pathways can substantially reduce the overall environmental impacts and resource use compared with conventional production. However, the benefits are only achieved when recovered resources can substitute virgin raw materials. If recovery simply adds new processing without reducing primary production, the environmental advantages are limited. This means that effective food resource recovery depends not only on technology but also on how systems are designed, regulated, and coordinated. Despite their promise, recovery systems face several challenges. Data on food losses and by-products remain incomplete and inconsistent, making it difficult to measure progress or prioritize actions. The quality and safety of recovered food resources can vary, and some promising technologies are still in early stages of development. Regulatory frameworks and public perceptions can also limit the use of recovered food resources, even when scientific evidence shows they are safe. Overcoming these barriers will require better data, supportive policy, and collaboration between industry, research, and government.

Nevertheless, the potential is immense. Large volumes of edible and near-edible food currently diverted to energy recovery or disposal could instead provide nutritious food and feed ingredients. Such circular strategies not only reduce waste and emissions but also enhance food security and preparedness by reducing dependence on imported raw materials. In a world of increasing resource scarcity and environmental constraints, transforming "waste" into a resource is both an environmental necessity and a sustainability opportunity.

In essence, this thesis shows that what we often consider as "wasted food" may, in fact, be among the most valuable resources for building the sustainable food systems of the future. By rethinking how we produce, consume, and value food, Sweden and the European Union can move toward food systems where no valuable resource is wasted, and where sustainability and resilience go hand in hand.

Populärvetenskaplig sammanfattning

Varje år går enorma mängder mat förlorad eller slängs. Globalt uppskattas nästan en tredjedel av all producerad mat aldrig nå våra tallrikar, och på europeisk nivå går omkring 14% av all mat förlorad redan innan den når butikshyllan. I Sverige kastas mer än en miljon ton fast mat varje år, av vilken mycket är fullt ätbar eller hade kunnat användas i nya livsmedels- eller foderprodukter. Samtidigt bidrar en ohållbar produktion och konsumtion av livsmedel till eskalerande påfrestningar för planeten, inte minst genom klimatutsläpp, förlust av biologisk mångfald och utarmning av naturresurser. Vi står alltså inför en kritisk utmaning: för att kunna försörja en växande befolkning inom planetens gränser måste vi använda den mat och de resurser vi redan har på ett bättre sätt.

Denna avhandling utvärderar miljönyttan av att möjliggöra återcirculering av överskottsmat och biprodukter, i syfte att stärka produktionen av mat och foder utan att öka behovet av råvaruutvinning. Istället för att betrakta dessa resurser som matsvinn eller livsmedelsförluster, utvärderas hur och under vilka förutsättningar de kan bli värdefulla råvaror i nya livsmedel och foder. Med hjälp av livscykelanalys (LCA) har flera återvinningsvägar studerats: från biokonvertering och genetisk förädling av livsmedelsrester till hantering av överskottsproduktion i bagerier och butiker, till återanvändning av dessa resurser i näringsbalanserat fiskfoder.

Resultaten visar tydligt att dessa återvinningsstrategier kan minska miljöpåverkan och öka resurseffektiviteten avsevärt jämfört med konventionell produktion. Men de miljömässiga vinsterna uppstår endast när resurserna faktiskt ersätter nyproducerade råvaror. Effektiv och miljömässigt hållbar resursåtervinning handlar därför inte bara om teknik, utan också om hur systemet utformas, regleras och samordnas. Samtidigt är möjligheterna mycket stora, då enorma mängder resurser idag inte används i enlighet med dess högsta potentiella värde. Sammanfattningsvis visar avhandlingen att det vi idag betraktar som ”matsvinn” i själva verket kan vara en av våra mest värdefulla resurser för att bygga framtidens hållbara livsmedelssystem. Genom att ompröva hur vi värderar dessa resurser kan Sverige och Europa ta steg mot ett livsmedelssystem där inga värdefulla resurser går till spillo, och där hållbarhet, resiliens, och miljömässig hållbarhet går hand i hand.

Acknowledgements

I would like to express my deepest gratitude to everyone who has supported and guided me throughout my PhD journey.

First and foremost, my sincere appreciation goes to my main supervisor, Mattias Eriksson, for your guidance, encouragement, and exceptional ability to turn complexity into clarity. I am equally grateful to my co-supervisors, Ingrid Strid and Pedro Brancoli, for your thoughtful advice, creative insights, and constructive feedback at every stage of this work. I could not have wished for a more engaged and supportive supervisory team. I am also sincerely grateful to my co-authors for stimulating collaborations and to all partners who contributed their knowledge and perspectives to this research.

To my wonderful roommates, Niina, Shan, and Lovisa, and to my extended *"food waste family"* — Christopher, Amanda, Sanna, and Joar: thank you for your friendship, endless encouragement, and conversations that never cease to inspire me. I would also like to express my deepest appreciation to all fellow PhD students who, over countless cups of coffee, made my days and work both fun and rewarding, and to my colleagues at the Department of Energy and Technology for inspiring discussions and shared enthusiasm for research. It has truly been a privilege to share this experience with such a talented and supportive community. My sincere thanks further go to the administrative staff for your kindness and steady support. I also gratefully acknowledge the financial support that made this research possible.

On a more personal note, I am deeply thankful to my friends and family for their love, positivity, and curiosity about my work and the world we share. Your support and optimism have propelled me forward, carrying me farther than I ever imagined.

Finally, to Hans — thank you for your love, patience, joy, and unconditional belief in me. Your encouragement gave me the confidence to follow my dreams and take a professional path I never anticipated. Without you, I would not be writing these words today. You have my deepest gratitude.

The work described in this thesis was funded by multiple research grants. For *Paper I*, the work was funded by the Swedish Research Council for Sustainable Development (FORMAS) via project H2020 ERA-net Cofund on Sustainable Food Production and Consumption (SUSFOOD2), grant number FR-2018/0001, the Finnish Ministry of Agriculture and Forestry, grant number 56/03.01.02/2018, and the German Federal Ministry of Food and Agriculture (BMEL), grant number 2818ERA07B.

For *Paper II*, the work was funded by the Swedish Research Council for Sustainable Development (FORMAS), and *Paper III* was funded by the European Union's Horizon 2020 - Research and Innovation Framework Programme under Grant Agreement no. 101000439, via the H2020 project LOWINFOOD (Multi-actor design of low-waste food value chains through the demonstration of innovative solutions to reduce food loss and waste).

For *Paper IV*, the work was funded by The Kamprad Family Foundation for Entrepreneurship, Research and Charity, grant no 20230075, and by the NordicFeed project (NordForsk 102761) and the ForestFeed project (Bioeconomy in the North, 344116).

Appendix

Table A1. Environmental impacts in *Paper I*, at the product level (**P**), alongside impacts of using this resource to substitute (food) or (feed). Expressed per kg DHA.

Midpoint level	**Unit**	Algae oil (**P**)	Fish oil (feed)
Global warming	kg CO_2 eq	7.5×10^{0}	-2.2×10^{1}
Stratospheric ozone depletion	kg CFC11 eq	1.1×10^{-5}	2.0×10^{-6}
Ionizing radiation	kBq Co-60 eq	2.5×10^{1}	2.5×10^{1}
Ozone formation, $_{\text{Human health}}$	kg NOx eq	1.7×10^{-2}	-1.1×10^{-1}
Fine particulate matter formation	kg PM2.5 eq	8.4×10^{-3}	-5.3×10^{-2}
Ozone formation, $_{\text{Terrestrial ecosystems}}$	kg NOx eq	1.8×10^{-2}	-1.1×10^{-1}
Terrestrial acidification	kg SO_2 eq	2.1×10^{-2}	-1.4×10^{-1}
Freshwater eutrophication	kg P eq	2.6×10^{-3}	-8.0×10^{-3}
Marine eutrophication	kg N eq	8.2×10^{-4}	1.1×10^{-4}
Terrestrial ecotoxicity	kg 1,4-DCB	6.4×10^{1}	-6.8×10^{1}
Freshwater ecotoxicity	kg 1,4-DCB	6.5×10^{-1}	-1.3×10^{-1}
Marine ecotoxicity	kg 1,4-DCB	8.5×10^{-1}	-4.1×10^{-1}
Human carcinogenic toxicity	kg 1,4-DCB	7.7×10^{-1}	-5.1×10^{-1}
Human non-carcinogenic toxicity	kg 1,4-DCB	1.1×10^{1}	-1.2×10^{1}
Land use	m^2a crop eq	2.1×10^{0}	7.3×10^{-1}
Mineral resource scarcity	kg Cu eq	5.5×10^{-2}	8.4×10^{-4}
Fossil resource scarcity	kg oil eq	1.8×10^{0}	-5.2×10^{0}
Water consumption	m^3	5.1×10^{-1}	2.0×10^{-1}
Endpoint level	**Unit**		
Human health	DALY	1.8×10^{-5}	-5.7×10^{-5}
Ecosystems	species × yr	5.3×10^{-8}	-1.0×10^{-7}
Resources	USD2013	6.2×10^{-1}	-1.5×10^{0}

Table A2. Environmental impacts in *Paper II*, at the product level (**P**), alongside impacts of using this resource to substitute (food) or (feed). Expressed per kg protein.

Midpoint level	**Unit**	Potato protein (**P**)	Eggs (food)	Fish meal (feed)
Global warming	kg CO_2 eq	2.5×10^{0}	-3.2×10^{0}	4.2×10^{-1}
Stratospheric ozone depletion	kg CFC11 eq	5.8×10^{-6}	-2.6×10^{-5}	5.2×10^{-6}
Ionizing radiation	kBq Co-60 eq	3.8×10^{-1}	3.0×10^{-1}	3.6×10^{-1}
Ozone formation, Human health	kg NOx eq	6.6×10^{-3}	-4.5×10^{-3}	-2.3×10^{-3}
Fine particulate matter formation	kg PM2.5 eq	3.6×10^{-3}	-9.9×10^{-3}	-7.5×10^{-4}
Ozone formation, Terrestrial ecosystems	kg NOx eq	6.8×10^{-3}	-4.5×10^{-3}	-2.3×10^{-3}
Terrestrial acidification	kg SO_2 eq	1.1×10^{-2}	-8.7×10^{-2}	-3.6×10^{-4}
Freshwater eutrophication	kg P eq	8.2×10^{-4}	-1.6×10^{-4}	6.2×10^{-5}
Marine eutrophication	kg N eq	4.0×10^{-4}	-7.5×10^{-3}	3.5×10^{-4}
Terrestrial ecotoxicity	kg 1,4-DCB	1.9×10^{1}	1.2×10^{1}	9.1×10^{0}
Freshwater ecotoxicity	kg 1,4-DCB	1.3×10^{-1}	-2.0×10^{-1}	7.8×10^{-2}
Marine ecotoxicity	kg 1,4-DCB	1.9×10^{-1}	1.1×10^{-1}	9.6×10^{-2}
Human carcinogenic toxicity	kg 1,4-DCB	2.2×10^{-1}	2.0×10^{-1}	1.3×10^{-1}
Human non-carcinogenic toxicity	kg 1,4-DCB	4.5×10^{0}	-2.6×10^{0}	2.9×10^{0}
Land use	m^2a crop eq	3.4×10^{-1}	-7.6×10^{0}	2.4×10^{-1}
Mineral resource scarcity	kg Cu eq	1.9×10^{-2}	1.6×10^{-2}	1.5×10^{-2}
Fossil resource scarcity	kg oil eq	7.5×10^{-1}	7.6×10^{-2}	2.5×10^{-1}
Water consumption	m^3	3.2×10^{-1}	1.7×10^{-1}	3.0×10^{-1}
Endpoint level	**Unit**			
Human health	DALY	7.1×10^{-6}	-8.6×10^{-6}	1.7×10^{-6}
Ecosystems	species × yr	1.9×10^{-8}	-9.3×10^{-8}	7.3×10^{-9}
Resources	USD2013	2.7×10^{-1}	1.9×10^{-2}	1.2×10^{-1}

Table A3. Environmental impacts in *Paper III*, at the product level (**P**), alongside impacts of using this resource to substitute (food) or (feed). Expressed per kg carbohydrate.

Midpoint level	**Unit**	Surplus bread (**P**)	Human nutrition (food)	Animal nutrition (feed)
Global warming	kg CO_2 eq	2.9×10^{0}	-2.8×10^{0}	2.2×10^{0}
Stratospheric ozone depletion	kg CFC11 eq	3.3×10^{-5}	-3.3×10^{-5}	2.6×10^{-5}
Ionizing radiation	kBq Co-60 eq	7.2×10^{-1}	-7.2×10^{-1}	7.0×10^{-1}
Ozone formation, $_{\text{Human health}}$	kg NOx eq	8.2×10^{-3}	-8.1×10^{-3}	5.8×10^{-3}
Fine particulate matter formation	kg PM2.5 eq	5.0×10^{-3}	-5.0×10^{-3}	3.4×10^{-3}
Ozone formation, $_{\text{Terrestrial ecosystems}}$	kg NOx eq	8.4×10^{-3}	-8.3×10^{-3}	5.9×10^{-3}
Terrestrial acidification	kg SO_2 eq	2.5×10^{-2}	-2.4×10^{-2}	1.9×10^{-2}
Freshwater eutrophication	kg P eq	1.5×10^{-3}	-1.5×10^{-3}	1.2×10^{-3}
Marine eutrophication	kg N eq	7.4×10^{-3}	-7.4×10^{-3}	5.7×10^{-3}
Terrestrial ecotoxicity	kg 1,4-DCB	1.4×10^{1}	-1.4×10^{1}	1.2×10^{1}
Freshwater ecotoxicity	kg 1,4-DCB	1.9×10^{-1}	-1.8×10^{-1}	1.6×10^{-1}
Marine ecotoxicity	kg 1,4-DCB	1.4×10^{-1}	-1.4×10^{-1}	1.2×10^{-1}
Human carcinogenic toxicity	kg 1,4-DCB	1.1×10^{-1}	-1.1×10^{-1}	8.3×10^{-2}
Human non-carcinogenic toxicity	kg 1,4-DCB	5.4×10^{0}	-5.3×10^{0}	4.8×10^{0}
Land use	m^2a crop eq	8.2×10^{0}	-8.2×10^{0}	6.8×10^{0}
Mineral resource scarcity	kg Cu eq	1.1×10^{-2}	-1.1×10^{-2}	8.2×10^{-3}
Fossil resource scarcity	kg oil eq	5.5×10^{-1}	-5.3×10^{-1}	4.1×10^{-1}
Water consumption	m^3	4.3×10^{-1}	-4.3×10^{-1}	2.3×10^{-1}
Endpoint level	**Unit**			
Human health	DALY	8.0×10^{-6}	-7.9×10^{-6}	6.0×10^{-6}
Ecosystems	species × yr	9.2×10^{-8}	-9.1×10^{-8}	7.5×10^{-8}
Resources	USD2013	2.0×10^{-1}	-1.9×10^{-1}	1.6×10^{-1}

Table A4. Environmental impacts in *Paper IV*, when using recovered resources to substitute (feed). Expressed per kg aquafeed.

Midpoint level	**Unit**	Algae scenario (feed)	Potato scenario (feed)	Bread scenario (feed)	Multi-source scenario (feed)
Global warming	kg CO_2 eq	-3.6×10^{-1}	-3.2×10^{-1}	-6.0×10^{-2}	-2.1×10^{0}
Stratospheric ozone depletion	kg CFC11 eq	-4.1×10^{-6}	-2.0×10^{-6}	2.1×10^{-6}	-5.0×10^{-6}
Ionizing radiation	kBq Co-60 eq	1.2×10^{0}	3.2×10^{-2}	6.8×10^{-2}	1.0×10^{0}
Ozone formation, $_{\text{Human health}}$	kg NOx eq	-1.6×10^{-3}	-4.7×10^{-4}	-4.6×10^{-4}	-4.0×10^{-3}
Fine particulate matter formation	kg PM2.5 eq	-8.4×10^{-4}	-4.8×10^{-4}	-4.9×10^{-4}	-2.0×10^{-3}
Ozone formation, $_{\text{Terrestrial ecosystems}}$	kg NOx eq	-1.6×10^{-3}	-4.6×10^{-4}	-4.7×10^{-4}	-4.1×10^{-3}
Terrestrial acidification	kg SO_2 eq	-4.1×10^{-3}	-1.4×10^{-3}	1.1×10^{-3}	-5.2×10^{-3}
Freshwater eutrophication	kg P eq	-8.9×10^{-5}	-4.9×10^{-5}	-8.3×10^{-5}	-3.6×10^{-4}
Marine eutrophication	kg N eq	-1.4×10^{-3}	-9.6×10^{-4}	3.5×10^{-4}	-2.0×10^{-3}
Terrestrial ecotoxicity	kg 1,4-DCB	-1.1×10^{-1}	1.4×10^{0}	-5.1×10^{-2}	-4.8×10^{-1}
Freshwater ecotoxicity	kg 1,4-DCB	-3.9×10^{-2}	-1.6×10^{-2}	-2.5×10^{-2}	-9.6×10^{-2}
Marine ecotoxicity	kg 1,4-DCB	1.3×10^{-2}	8.0×10^{-3}	-9.0×10^{-3}	4.6×10^{-3}
Human carcinogenic toxicity	kg 1,4-DCB	2.0×10^{-2}	-2.2×10^{-2}	-6.3×10^{-2}	-4.5×10^{-2}
Human non-carcinogenic toxicity	kg 1,4-DCB	-7.1×10^{-1}	-8.9×10^{-1}	-1.2×10^{-1}	-1.9×10^{0}
Land use	m^2a crop eq	-7.4×10^{-1}	-2.6×10^{-1}	3.8×10^{-1}	-2.3×10^{0}
Mineral resource scarcity	kg Cu eq	1.7×10^{-3}	8.9×10^{-4}	-1.3×10^{-3}	5.3×10^{-4}
Fossil resource scarcity	kg oil eq	-4.4×10^{-2}	-6.2×10^{-2}	-4.1×10^{-2}	-1.8×10^{-1}
Water consumption	m^3	2.0×10^{-2}	-4.8×10^{-2}	-1.2×10^{-1}	-1.8×10^{-1}
Endpoint level	**Unit**				
Human health	DALY	-9.0×10^{-7}	-6.9×10^{-7}	-5.5×10^{-7}	-3.7×10^{-6}
Ecosystems	species × yr	-8.6×10^{-9}	-3.4×10^{-9}	2.7×10^{-9}	-2.9×10^{-8}
Resources	USD2013	-1.3×10^{-2}	-7.6×10^{-3}	7.1×10^{-3}	-4.0×10^{-2}

Sustainable Production and Consumption 27 (2021) 2002–2021

Contents lists available at ScienceDirect

Sustainable Production and Consumption

journal homepage: www.elsevier.com/locate/spc

Research article

Life cycle assessment of fish oil substitute produced by microalgae using food waste

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A R T I C L E I N F O

Article history:
Received 2 February 2021
Revised 29 April 2021
Accepted 30 April 2021
Available online 8 May 2021

Editor: Prof. Adisa Azapagic

Keywords:
Life cycle assessment (LCA)
Volatile fatty acids (VFA)
Docosahexaenoic acid (DHA)
Biodiversity
Food waste

A B S T R A C T

Fish oil has been used in conventional aquaculture for decades, despite the known links between increasing global demand for fish and depletion of natural resources and vital ecosystems (FAO, 2020, 2019). Alternative feed ingredients, including algae oil rich in *docosahexaenoic acid* (DHA), has therefore been increasingly used to substitute traditional fish oil. Heterotrophic algae cultivation in bioreactors can be supported by a primary carbon feedstock recovered from food waste, a solution that could reduce environmental impacts and support the transition towards circular food systems. This study used life cycle assessment to quantify environmental impact of DHA produced by the heterotrophic algae *Crypthecodinium cohnii,* using short-chain carboxylic acids derived from dark fermentation of food waste. The future potential of DHA from algae was evaluated by comparing the environmental impact to that of DHA from Peruvian anchovy oil. With respect to global warming, terrestrial acidification, freshwater eutrophication and land use, algae oil inferred -52 ton CO_2eq, 3.5 ton SO_2eq, -94 kg Peq, 2700 m^2 eq, respectively per ton DHA. In comparison, the impact per ton DHA from fish oil was -15 ton CO_2eq, 3.9 ton SO_2eq, -97 kg Peq and 3200 m^2 eq. Furthermore, algae oil showed lower climate impact compared to canola and linseed oil. By including Ecosystem damage as indicator for ecosystem quality at endpoint level, the important aspect of biodiversity impact was accounted for. Although the method primarily accounts for indirect effects on biodiversity, DHA from algae oil showed lower Ecosystem damage compared to fish oil even when future energy development, optimized production, increased energy demand and effects on biotic resources were considered via sensitivity analyses. As the results suggest, algae oil holds a promising potential for increased sustainability within aquaculture, provided that continued development and optimization of this emerging technology is enabled through active decision-making and purposeful investments.

1. Introduction

Today, aquaculture is the fastest growing food-producing sector, originally developed to support an increasing global population with nutritious food and essential Omega-3 fatty acids (FAO, 2018). Fish oil has been one of the most important ingredients in conventionally produced aquaculture feed for decades, primarily since it contains bioavailable polyunsaturated fatty acids, including the 6-fold unsaturated fatty acid *docosahexaenoic acid* (DHA). The original aim in farming high-value fish was to preserve marine biodiversity while supporting an increased global demand for food fish (European Commission, 2019). However, current fish oil supply is highly dependent on fossil energy and marine raw materials, and research has shown that natural resources and ecosystems are being depleted as the global demand for fish increases (FAO, 2020, 2019).

In aquatic ecosystems, DHA is naturally produced by planktonic microalgae and is accumulated in fish via the food web (Colombo et al., 2020; Sprague et al., 2017). As DHA cannot be synthesised by animals, a sufficient intake must be obtained through the diet. Therefore, fish oil rich in DHA is often added in food and feed production to enhance nutrition levels in dairy, meat and fish consumed by humans (Silva et al., 2018; Toppe, 2013). Each year, around 1 million tons of fish oil is used to produce aquafeed (Beal et al., 2018), where Peruvian anchovy oil contains one of the highest concentrations of DHA (IFFO, 2017). Conventional fish oil production depends on marine raw materials, primarily for-

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https://doi.org/10.1016/j.spc.2021.04.033

age fish such as anchovy, and finite resources such as fossil energy to power the conversion processes (Rodríguez et al., 2019; Tyedmers et al., 2006). Fish oil production has also been shown to contribute to two of the most urgent threats to life on Earth: global warming and loss of biodiversity (Center for Biological Diversity, 2020; Ghamkhar and Hicks, 2020; United Nations, 2019). Moreover, due to global warming the natural DHA synthesis by marine microalgae is predicted to decrease by 58% until 2100 (Colombo et al., 2020), which would reduce the natural DHA content in fish oil. In addition, long-term monitoring of marine fish stocks by FAO (2020) showed that 34% of fish stocks were overfished in 2017. Thus, the dependency on fish oil for aquafeed must be reduced to ensure future food security (Cottrell et al., 2020) and to achieve sustainable market growth of aquaculture (Hardy, 2010).

New production methods with lower environmental impact are needed to maintain vital ecosystems, increase resource efficiency and support future sustainability within food supply chains. A potential solution to the impact associated with conventional fish oil production could be to gain DHA directly from the marine primary producers, namely microalgae. Using microalgae as a novel aquafeed could decrease the demand for forage fish in aquaculture, while still maintaining the required DHA profile for high-value aquaculture fish feed (Beal et al., 2018; Sprague et al., 2017, 2015). Previous research has established that heterotrophic microalgae can be cultivated in bioreactors using carbon sources derived from food waste as primary feedstock (Chalima et al., 2020, 2019, 2017), thus enabling an alternative resource recovery solution within food waste management.

Today, about 1.3 billion ton food is wasted globally each year, of which over 900 million ton origin from households, retail establishments and the food service sector (UNEP, 2021). Food waste accounts for a considerable proportion of environmental burden for the current food supply chain, especially since current practice are often based on a linear production process (Laso et al., 2018). Increased resource recovery and nutrient recycling are considered key actions to achieve long-term sustainability, especially within the future food system and food waste management. Life cycle assessment (LCA) is a method commonly used to systematically assess and quantify the environmental impact for a process or product. By considering resource use and emissions related to the whole production chain, LCA is a valuable tool for evaluating if a suggested solution can reduce the environmental impact compared with a reference scenario. The aim of this study was to evaluate the future potential of DHA produced from algae with substrate originating from DF using food waste, by assessing and comparing the environmental impact to that of DHA produced from Peruvian anchovy oil. A quantitative assessment of the environmental impact and the potential effects on biodiversity was included to provide a vital dimension of aquaculture and food waste valorisation to policymakers, the research community and the industry. The long-term goal with this study was to support sustainable development, by assessing innovative solutions for food waste valorisation within a circular economy approach.

2. Literature review

The environmental impact of conventional fish oil production has been thoroughly studied by previous research (Avadí and Fréon, 2013; Fréon et al., 2017; Silva et al., 2018). To increase sustainability within aquaculture and reduce dependency on traditionally used marine raw material, alternative DHA sources in aquafeed has gained a lot of scientific attention (Bélanger-Lamonde et al., 2018; Glencross et al., 2020). Vegetable sources, such as canola or linseed oil, require conversion of *alpha lipoic acid* (ALA) via digestion to provide bioavailable DHA (Kannan et al., 2021; Russo et al., 2021). Although often considered controversial, genetically modified crops has been shown to increase DHA potential in plants to levels comparable with fish oil (Petrie et al., 2020; West et al., 2021; Zhou et al., 2019). An alternative source that provide bioavailable DHA are yeast oil and algae oil (Parsons et al., 2019; Porcelli et al., 2020; Yarnold et al., 2019). Research suggest that microbial oils from heterotrophic algae and yeast will likely be common within future food systems (Parsons et al., 2018), but microalgae oils are considered one of the most promising future fish oil substitutes (Cottrell et al., 2020; Oliver et al., 2020). Both total and partial replacement of fish oil in aquafeed is currently possible, but the nutritional composition of the substituting oil must be considered to maintain the required nutritional value in food fish (Sarker et al., 2016; Schade et al., 2020; Sissener, 2018). Ghamkhar and Hicks (2020) concluded that sole fish oil replacement can infer reduced stress on biotic resources, but they also highlighted that technologies for producing substituting oils needs further improvement to achieve more effective energy use and mitigate potential burden shifts. Commercial algae derived products, including DHA for feed and food, are well established on the market and will likely be further developed to improve effectiveness and reduce production costs (Patel et al., 2020a; Yarnold et al., 2019).

Thraustochytriaceae and the dinoflagellate *Cryptecodinium* sp. are eukaryotic, planktonic heterotrophic marine microalgae that are known to accumulate DHA to beyond 10% (w/w) of their cell weight (Kumar et al., 2021; Mendes et al., 2009). The intensively examined strain *Cryptecodinium cohnii* (*C. cohnii*) is cultivated aerobically in bioreactors supported by a carbon source as primary feedstock. Different feedstock has been frequently used, including glucose (Deprá et al., 2020), lignocellulosic biomass (Karnaouri et al., 2020) and olive pomace (Paz et al., 2020). Acetic acids and other short-chain fatty acids has been shown to greatly benefit microalgae DHA accumulation (Hillig, 2014; Sijtsma et al., 2010), especially waste derived *volatile fatty acids* (Oliver et al., 2020). Volatile fatty acids (VFA) are obtainable from anaerobic digestion (AD), in particular dark fermentation (DF) (Chalima et al., 2019; Fei et al., 2015; Patel et al., 2020a). DF is a common food waste (FW) valorisation process where the first two steps, hydrolysis and acidogenesis of the typical AD, yield short-chain carboxylic acids (mostly VFAs). VFAs extracted from AD or DF primarily consist of acetic acid ($C_2H_4O_2$) and smaller amounts of propionic acid ($C_3H_6O_2$) and butyric acid ($C_4H_8O_2$), while the remaining biomass can still be used to produce biogas (Kim et al., 2019; Tampio et al., 2019). Previous studies by Paritosh et al. (2017) and Wainaina et al. (2019) suggest that up to 20 g VFA/L FW can be produced in the hydrolytic/ acidogenesis stage of conventional two-step AD, while up to 25 g VFA/L FW was suggested by Herrero Garcia et al. (2018) during optimised DF. Multiple studies have emphasised the prosperous potential for using VFA recovered from fermented FW as a sustainable alternative to conventional carbon sources for bioproduction of DHA from microalgae. The synthesis of valuable by-products like squalene can increase the worth of heterotrophic algae biomass (Patel et al., 2020b). Chalima et al., (2017) reviewed how VFA could be reused in a microalgae fermentation process, and a later study also included the VFA separation from DF and their use for the bioproduction to high added-value DHA by *C. cohnii* (Chalima et al., 2020, 2019). Patel et al. (2021) and Fei et al. (2015) both studied the effects on fatty acid accumulation when using VFA as primary carbon feedstock in heterotrophic algae cultivation. They found that a similar fatty acid accumulation could be obtained with VFA as when using traditional glucose, but the effect was largely affected by the amount and ratio of VFAs. Combining bioconversion of VFA to DHA through microalgae in combination with energy generation from FW, can infer increased sustainability within aquaculture (Oliver et al., 2020) and energy production (Chalima et al., 2019; Paritosh et al., 2017). This approach also enables a novel solution for FW resource recovery, which has been widely recognised as an important component in sustainable development within the food system (Brancoli et al., 2020; Scherhaufer et al., 2020; Teigiserova et al., 2020). As suggested by Woodhouse et al. (2018), primary production (including aquaculture and fishing) are generally main environmental hotspots in the food supply chain. Using a more circular flow of resources might reduce the burden from primary production, especially when considering fish oil substitutes. Current research suggest that microalgae will likely play an important role in maintained global food security by bridging a future gap between supply and demand for DHA (Jovanovic et al., 2021; Russo et al., 2021; Tocher et al., 2019). To ensure global food security and maintain natures ability to provide resources in the future, substantial efforts are required to develop new production methods with lower environmental impact that also comply with the global Agenda 2030 and sustainable development goals (SDG) (Herrero et al., 2020; United Nations, 2020, 2019).

Identifying and understanding the environmental implications of new technologies is thus crucial to ensure that suggested new solutions also support future sustainability. Multiple studies have assessed the environmental impact of algae oil intended for the food supply chain (Beal et al., 2018; Porcelli et al., 2020; Schade et al., 2020). To date, previous LCA studies have mostly focused on phototrophic algae processes (Barr and Landis, 2018; Keller et al., 2017), where required inputs and outputs differ substantially from those in heterotrophic cultivation (Smetana et al., 2017). Similarly, assessments of algae oil used for biofuel production is more common than algae oil intended for food or feed (Hosseinzadeh-Bandbafha et al., 2020; Shi et al., 2019). However, Rösch et al. (2019) suggests that the inputs required for algae cultivation and harvest are often similar regardless of how the end product is used. In a recent study, Deprá et al. (2020) assessed environmental impacts with respect to commercial microalgae-based products, including DHA produced from *C. cohnii*. They concluded that microalgae as a source for DHA owns a high sustainable potential. Although commercialized in recent years, to our knowledge, no LCA study has yet assessed environmental impact of large-scale heterotrophic algae oil production intended to substitute fish oil.

Some of the most frequently included impact categories for seafood LCA are global warming, acidification and eutrophication (Ruiz-Salmón et al., 2021). Scherer et al. (2020) further suggest to also include land and sea use impact when addressing food security and biodiversity conservation, an aspect that alongside removal of fish stocks also has been highlighted by Langlois et al. (2015) and Hélias et al. (2018). Climate change, nutrient pollution, change in habitat, overexploitation and invasive species are so-called drivers of biodiversity loss that provide a measurable link between human actions and ecosystem damage (Watson et al., 2005). Even though frequently identified as especially important for LCA with a marine food focus, few previous studies have accounted for the impacts related to biotic resource use due to its complexity (Marques et al., 2021; Scherer et al., 2020; Winter et al., 2017). Following the cause-effect-chain in LCA, impact at midpoint infer damage to an area of protection at endpoint level. Impact at midpoint level causing loss of biodiversity infer damage to ecosystem quality, which in turn lead to Ecosystem damage. Biotic impact indicators are still under development and are therefore currently not included in commercial LCA methods (Crenna et al., 2020; Marques et al., 2021; Ruiz-Salmón et al., 2021). As suggested by Asselin et al. (2020), standardised LCA methods currently only cover three of the five identified drivers for biodiversity loss, while overexploitation and invasive species still need to be further developed. However, climate change, pollution and change in habitat at midpoint level can all be linked to ecosystem quality and expressed in a common endpoint unit (Crenna et al., 2020; Huijbregts et al., 2016; Woods et al., 2016). Moreover, there is an urgent need improved *life cycle inventory* (LCI) data to fully evaluate the sustainability of microalgae cultivation (Avadí et al., 2020; Lopes da Silva et al., 2019).

3. Material and method

3.1. Goal and scope

The LCA method (ISO, 2006a, 2006b) was used to assess environmental impact of fish oil substitute produced by microalgae using VFA derived from food waste as primary feedstock. By using an attributional (ALCA) approach, the aim was to identify resource-demanding flows and provide results via *life cycle impact assessment* (LCIA) that could support innovations within future DHA production and food waste management. The technology for large scale algae oil production using VFA from dark fermentation of food waste is currently in the development phase. To assess the system, laboratory and full-scale input and output data from previous studies was compiled and used to model a large scale DHA production. A physical functional unit (1 ton DHA) was selected to also reflect the nutritional function of the product, as proposed by McAuliffe et al. (2020). Data from Ecoinvent 3.5 were used for the background system (see **Table A.1** in Appendix A), while substitution via system expansion was used in the foreground system to allocate the environmental burden between the two main by-products electricity and heat. System expansion was favoured to avoid economic or mass allocation, as suggested by the ISO 14040 -series (ISO, 2006a).

3.2. Description of scenarios

The studied systems were modelled as two parallel scenarios to assess large-scale production of DHA: a conceptual *Algae scenario*, where DHA was produced from *C. cohnii* microalgae using VFA from DF with food waste, and a conventional *Fish scenario*, where DHA was derived from Peruvian anchovy (Fig. 1). Included in the system boundary were production and end-of-life for required inputs, as well as construction of buildings and energy used for processing. Construction and maintenance of additional infrastructure were outside the scope of this study. Transport was included for inputs and outputs, while intermediate transport at the production site was excluded. NTMCalc Basic 4.0 was used to estimate transport distances for inputs, where a freight lorry of Euro class VI was assumed for most road transport.

The site location for algae oil and energy production was assumed to be Berlin, Germany, while fish oil production was assumed to be located in Lima, Peru. Site-specific data were primarily used, so the results are site dependent. The available food waste was considered a free resource and thereby did not contribute to the environmental impact, since production belongs to the preceding food system. The digestate was assumed to have a negligible market value in comparison with biogas and biohydrogen, therefore only transport from the AD plant was included, and not the end-of-life for digestate. Since the energy source strongly affects LCA results, a customised dataset for electricity and heat input and output was created using data from 2020 for the German electricity and heat production mix (see Table B.1 in Appendix B). Heat produced in $Energy_{DF}$ and $Energy_{AD}$ was assumed to be re-used during algae oil production, DF and AD. Avoided electricity due to biogas production was assumed to replace equal parts of the fossil energy sources lignite, natural gas and coal, while residual avoided heat was assumed to replace natural gas.

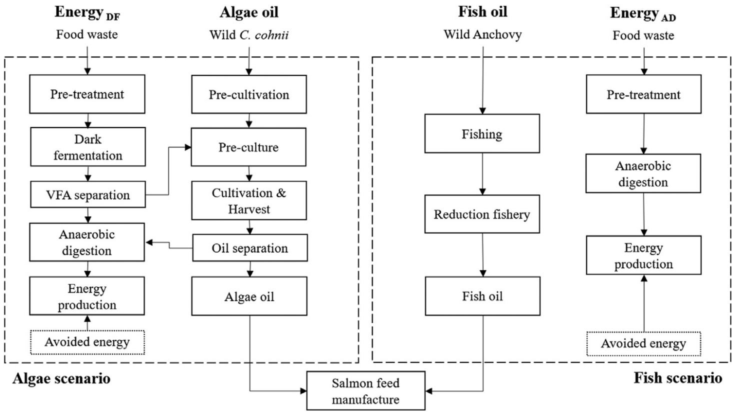

Fig. 1. Illustration of the model set-up for (left) the *Algae scenario* and (right) the *Fish scenario*. The dashed line represents the system boundary and the dotted line illustrates by-products included via system expansion. AD: anaerobic digestion, DF: dark fermentation, VFA: volatile fatty acids.

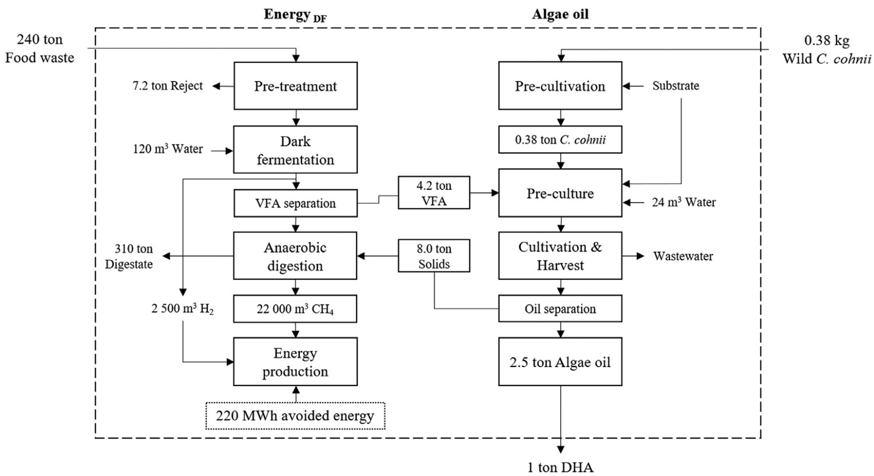

Fig. 2. Illustration of the model set-up for the *Algae scenario*. The dashed line represents the system boundary and the dotted line illustrates electricity and heat produced from biohydrogen and biogas, included via system expansion. Additional electricity, heat and transports are not included in this illustration.

3.2.1. Algae scenario

The *Algae scenario* (Fig. 2) process comprised production of *algae oil* via heterotrophic cultivation of *C. cohnii*, using VFA extracted from DF as the primary feedstock, and $Energy_{DF}$ that represent energy production using the remaining food waste and algae biomass to produce biohydrogen and biogas.

C. cohnii are grown in a pre-cultivation process that requires an algae biomass stock, nutrient substrate, water and electricity as inputs. The substrate required in heterotrophic cultivation of *C. cohnii* mainly consist of a carbon source, a nitrogen source for the first production phase, and salts (Mendes et al., 2009). The production process is usually divided into an initial phase of cell growth and a subsequent phase of nitrogen limitation, during which DHA is accumulated inside the cell. Adding VFA can increase the DHA yield, especially if fed to the second cultivation phase. 4.2 ton of VFA was required to produce 1 ton of DHA during cultivation, which was considered the limiting factor in this study. Input data for pre-cultivation were assumed to be similar as heterotrophic cultivation in closed fermenters suggested by Smetana et al. (2017), with a fishing dataset representing wild *C. cohnii* biomass from the ocean. All nutrient inputs were assumed to be transported by road to the bioreactor facility.

The substrate input for the pre-culture, including yeast and salt, to heterotrophic cultivation was used according to data from previous research (Deprá et al., 2020; Patel et al., 2021; Smetana et al., 2017). Less than 1% of initial *C. cohnii* biomass was used in the pre-culture, which after harvest and dewatering was assumed to make up 20% of the algae culture (200g/L DCW), which is considered the upper limit of reachable values. Since the carbon feedstock was primarily used for biomass growth, the amount of CO_2 emitted from heterotrophic algae cultivation was accounted for by assuming 0.1 kg CO_2 emissions to air per kg algae (Lopes da Silva et al., 2019). As stated by Keller et al. (2017) and Beal et al. (2018), a considerable fraction of water and residues are re-usable. There-

Table 1
Process data for *algae oil* production, expressed per 1 ton DHA.

		Amount	Unit	Ecoinvent dataset [E]
Pre-cultivation				
Input	Wild *C. cohnii*	3.8×10^{-1}	kg	1
	N fertiliser	7.5×10^{-2}	kg	2
	P fertiliser	3.8×10^{-2}	kg	3
	Glucose	1.3×10^{1}	kg	4
	Water	3.8×10^{-1}	m^3	5
	Natural gas	7.5×10^{-2}	kWh	6
	Steam	1.9×10^{1}	kg	7
	Electricity	6.8×10^{0}	kWh	(1)
	Processing facility	4.7×10^{-7}	unit	8
	Transport (substrate)	4.0×10^{-1}	tkm	9
Output	*C. cohnii*	3.8×10^{1}	kg	
	CO_2 to air	3.8×10^{0}	kg	10
Pre-culture				
Input	*C. cohnii*	3.8×10^{-2}	ton	
	VFA from DF	4.2×10^{0}	ton	
	Yeast	7.5×10^{-2}	ton	11
	Reef salt	9.4×10^{-1}	ton	12
	Molasses	3.4×10^{-1}	ton	13
	Water	2.4×10^{1}	m^3	5
	Processing facility	9.3×10^{-4}	unit	8
	Transport (substrate)	5.6×10^{3}	tkm	14
Output	Algae culture	5.4×10^{1}	ton	
Cultivation & Harvest				
Input	Algae culture	5.4×10^{1}	ton	
	Electricity	6.6×10^{0}	MWh	(1)
	Heat (re-used)	8.2×10^{-1}	MWh	
	Processing facility	9.3×10^{-4}	unit	8
Output	CO_2 to air	1.1×10^{0}	ton	10
	Algae suspension	1.1×10^{1}	ton	
	Wastewater	1.8×10^{1}	m^3	15
Oil separation				
Input	Algae suspension	1.1×10^{1}	ton	
	Electricity	5.5×10^{0}	MWh	(1)
	Heat (re-used)	9.6×10^{-3}	MWh	
	Processing facility	2.2×10^{-6}	unit	16
Output	Algae oil	2.5×10^{0}	ton	
	Algae solids	8.0×10^{0}	ton	
Transport to Norway				
Input	Transport (algae oil)	4.7×10^{3}	tkm	14
Output	DHA (algae oil)	1.0×10^{0}	ton	

[E]Ecoinvent datasets used in SimaPro, see Appendix A and B.

fore, a 50% recirculation of wastewater from cultivation and harvest was assumed in this study. Inputs for electricity and heat used for cultivation, harvest, dewatering and oil separation were based on previous studies (Hosseinzadeh-Bandbafha et al., 2020; Shi et al., 2019; Smetana et al., 2017). The oil separation process involves cell disruption using an oil mill, while additional electricity is used for mechanical pressing and heat was used for pre-treatment, extraction and drying of biomass (Passell et al., 2013). Mechanical pressing was favoured over chemical means to extract the DHA-containing oleaginous fraction from the biomass since the algae oil was intended for food (Lopes da Silva et al., 2019). To calculate the amount of algae required per functional unit, a 40% DHA content in algae oil were assumed alongside a biomass-specific DHA content of 10% in *C. cohnii* (Hillig et al., 2014; Swaaf et al., 2001). The lipid content in *C. cohnii* was set to a maximum of 24% (w/w), as suggested in previous studies (John, 2009; Mendes et al., 2009; Passell et al., 2013). The remaining solid algae biomass was re-used in AD to produce biogas. All inputs and outputs are presented in Table 1.

Energy$_{DF}$ represents the energy production process via DF with VFA extraction and anaerobic digestion (Fig. 2), during which biogas with 60% CH_4 content and biohydrogen was assumed to be produced. The amount of extractable VFA from DF was set to 18 g VFA/kg FW, which is similar to the AD output suggested by Paritosh et al. (2017). Assuming 97% biodegradable content in collected food waste, 240 ton were needed to produce 4.2 ton VFA. The 3% of non-biodegradable material was sorted out at pre-treatment and assumed to be equal parts of plastic that was re-used in municipal waste incineration and aluminium for recycling. The values applied for total solids (TS) and volatile solids (VS) in pre-treated food waste were 26% TS and 24% VS, as suggested by Yi et al. (2014) and Slorach et al. (2019). Electricity requirement in pre-treatment was set to 150 kWh/ton TS (Carlsson, 2015; Pöschl et al., 2010). Construction of the waste preparation facility was included. The total amount of electricity and heat required to produce biogas was similar to that in Opatokun et al. (2017), but it was assumed that 90% of total energy consumption was used during DF (mixing and pumping) and the remaining 10% in AD. Biohydrogen production potential of 45 m^3/t VS during DF was assumed, which is an average value based on previous studies (Hou et al., 2020; Pu et al., 2019; Wainaina et al., 2020). The processing unit used for DF and AD was assumed to be an AD plant with methane recovery. Data from a vegetable oil refinery were used to assess the VFA separation process, where 0.21 kWh electricity per kg extracted VFA was assumed (Hosseinzadeh-Bandbafha et al., 2020). Solids from algae production were re-used in AD and assumed to have equivalent CH_4 potential to the food waste effluent from DF, although the true value might be higher. The energy production process included energy and processing facilities for 40% electricity and 50% heat (Hakawati et al., 2017) from biogas, when assuming

Table 2
Process data for Energy$_{DF}$ production, expressed per 1 ton DHA.

Pre-treatment	Amount		Unit	Ecoinvent dataset [E]
Input	Food waste (FW)	2.4×10^2	ton	
	Electricity	9.4×10^0	MWh	(1)
	Waste prep. facility	4.8×10^{-4}	unit	17
	Transport (FW)	7.2×10^3	tkm	18
	Transport (reject)	2.2×10^2	tkm	14
Output	Pre-treated FW	2.3×10^2	ton	
	Metal reject	3.6×10^0	ton	19
	Plastic reject	3.6×10^0	ton	20
Dark fermentation				
Input	Pre-treated FW	2.3×10^2	ton	
	Water	1.2×10^2	m^3	5
	Heat (re-used)	6.6×10^0	MWh	
	Electricity	2.1×10^3	MWh	(1)
	Processing facility	1.9×10^{-3}	unit	21
Output	Slurry from DF	4.7×10^2	ton	
	Hydrogen (H_2)	2.5×10^3	m^3	
VFA extraction				
Input	Slurry from DF	4.7×10^2	ton	
	Electricity	8.8×10^{-1}	MWh	(1)
	Oil processing facility	4.6×10^{-7}	unit	22
Output	VFA	4.2×10^0	ton	
	Effluent from DF	3.4×10^2	ton	
Anaerobic digestion				
Input	Effluent from DF	3.4×10^2	ton	
	Algae solids	8.0×10^0	ton	
	Heat (re-used)	3.5×10^0	MWh	
	Electricity	2.4×10^{-1}	MWh	(1)
	Processing facility	1.0×10^{-2}	unit	21
	Transport (algae solids)	8.0×10^1	tkm	9
	Transport (digestate)	6.1×10^3	tkm	14
Output	Methane (CH_4)	2.2×10^4	m^3	
	Digestate	3.1×10^2	ton	
Energy production				
Input	Hydrogen (H_2)	2.5×10^3	m^3	
	Methane (CH_4)	2.2×10^4	m^3	
	CHP facility	1.0×10^2	unit	23
Output	Electricity	1.0×10^2	MWh	
	Heat	1.2×10^2	MWh	
	Heat (internal use)	1.1×10^1	MWh	
System expansion				
	Avoided electricity	1.0×10^2	MWh	(2)
	Avoided heat	1.2×10^2	MWh	24

[E]Ecoinvent datasets used in SimaPro, see Appendix A and B.

a 90% overall efficiency rate in combined heat and power (CHP) production. The input was calculated based on the amount of electricity generated. Standard values for the energy content in H_2 and CH_4 were used (2.99 and 9.97 kWh/m^3, respectively). Digestate production was calculated assuming 50% water re-circulation in AD. The main output was 1 ton DHA from algae oil, while the by-products electricity and heat from biohydrogen and biogas were included via system expansion as avoided energy. Inputs and outputs are presented in Table 2.

3.2.2. Fish scenario

The process for the *Fish scenario* (Fig. 3) comprised production of *fish oil* in reduction fisheries using wild Peruvian anchovy, and *Energy$_{AD}$* describing energy production using the same amount of food waste as was available in the *Algae scenario* to produce biogas in Germany. A 10% DHA content was assumed for Peruvian anchovy oil (Sissener, 2018). The amount of wild anchovy required to produce 10 ton fish oil was based on data in Silva et al. (2018). Transport from Lima to Norway via the harbour in Venezuela was included. Inputs and outputs are presented in Table 3.

Production of Energy$_{AD}$ assumed the same inputs and processing calculations as described for Energy$_{DF}$, but without DF, VFA extraction and addition of algae solids. Table 4 illustrate the process data used for Energy$_{AD}$ production. The main output from this scenario was 1 ton DHA from fish oil, while the by-product biogas (subsequently used for electricity and heat) was included via system expansion.

3.3. Life cycle impact assessment

Life cycle impact assessment translates emissions and resource use to environmental impact at either midpoint or endpoint level for selected impact categories. The midpoint and endpoint approaches are complementary, but the midpoint approach has lower modelling uncertainty and a stronger relation to the environmental impact, while the endpoint approach provides a better indication of the environmental relevance (Huijbregts et al., 2016). In general, characterisation at midpoint has lower uncertainty and a stronger relation to the elementary flows than endpoint characterization, but characterization at endpoint can provide a better indication of the environmental relevance of the flows. The ReCiPe 2016 method includes 18 midpoint categories that can be directly translated into endpoint impact, using a constant characterisation factor (CF) for each impact category. This is done by assuming that all stressors are identical after midpoint impact. The endpoint indicators can be aggregated into a common unit to describe the impact for an area of protection, e.g., Ecosystem damage that consists of 13 endpoint indicators expressed in the unit *species per*

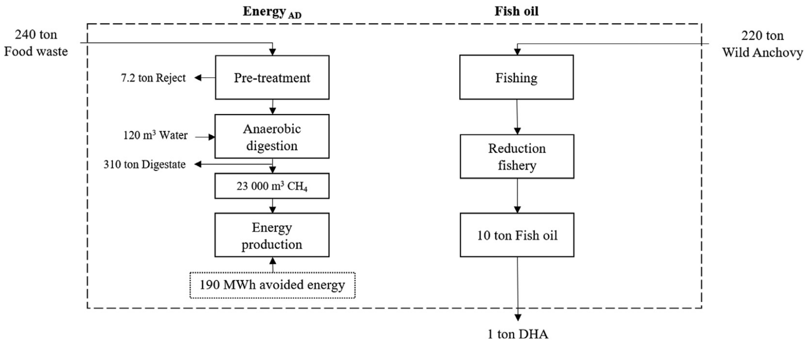

Fig. 3. Illustration of the model set-up for the *Fish scenario*. The dashed line represents the system boundary and the dotted line illustrates the electricity and heat produced from biogas, included via system expansion. Additional electricity, heat and transport are not included. AD: anaerobic digestion, DHA: docosahexaenoic acid.

Table 3
Process data for *fish oil* production, expressed per 1 ton DHA.

		Amount	Unit	Ecoinvent dataset [E]
Fishing				
Input	Wild anchovy	2.2×10^2	ton	
	Fishing process	2.1×10^2	ton	25
Output	Landed anchovy	2.1×10^2	ton	
	Discarded fish	8.7×10^0	ton	
Reduction fishery				
Input	Landed anchovy	2.1×10^2	ton	
	Processing facility	1.0×10^1	ton	26
Output	Fish oil	1.0×10^1	ton	
Transport to Norway				
Input				
	Transport (fish oil)	4.3×10^4	tkm	27
		8.1×10^4	tkm	28
		2.3×10^3	tkm	14
Output	DHA (fish oil)	1.0×10^0	ton	

[E]Ecoinvent datasets used in SimaPro, see Appendix A.

Table 4
Process data for $\text{Energy}_{\text{AD}}$ production, expressed per 1 ton DHA.

		Amount	Unit	Ecoinvent dataset [E]
Pre-treatment				
Input	Food waste (FW)	2.4×10^2	ton	
	Electricity	9.4×10^0	MWh	(1)
	Waste prep. facility	4.8×10^{-4}	unit	17
	Transport (FW)	7.2×10^3	tkm	18
	Transport (reject)	2.2×10^2	tkm	14
Output	Pre-treated FW	2.3×10^2	ton	
	Metal reject	3.6×10^0	ton	19
	Plastic reject	3.6×10^0	ton	20
Anaerobic digestion				
Input	Pre-treated FW	2.3×10^2	ton	
	Water	1.2×10^2	m^3	5
	Heat (re-used)	3.7×10^1	MWh	
	Electricity	2.3×10^0	MWh	(1)
	Processing facility	1.1×10^{-2}	unit	21
	Transport (digestate)	6.2×10^3	tkm	14
Output	Methane (CH_4)	2.3×10^4	m^3	
	Digestate	3.1×10^2	ton	
Energy production				
Input	Methane (CH_4)	2.3×10^4	m^3	
	CHP processing facility	1.0×10^2	unit	23
Output	Electricity	1.0×10^2	MWh	
	Heat	9.0×10^1	MWh	
	Heat (internal use)	3.7×10^1	MWh	
System expansion				
	Avoided electricity	1.0×10^2	MWh	(2)
	Avoided heat	9.0×10^1	MWh	24

[E]Ecoinvent datasets used in SimaPro, see Appendix A and B.

year (species.yr). This unit can be interpreted as the potential disappeared fraction of species each year. Species in this method include plants and microorganisms that support higher terrestrial and aquatic tropic levels of the food chain (Crenna et al., 2020). Loss of these species will affect the food chain, but the direct disappearance of higher organisms or the impact on endangered species are not included in the ReCiPe method. As concluded in a review by Ruiz-Salmón et al. (2021), the most frequently assessed impact categories for seafood at midpoint are global warming, acidification and eutrophication, while Scherer et al. (2020) also emphasise the importance of including land use and sea use impact when addressing food security and biodiversity conservation. These indicators (climate change, pollution and change in habitat) has also been identified as key contributors to biodiversity loss and damage to ecosystem quality at endpoint level (Díaz et al., 2019; Marques et al., 2021; Scherer et al., 2020). Therefore, this study assesses global warming, terrestrial acidification, freshwater eutrophication and land use at midpoint level. To enable a more holistic assessment, which is especially important for systems dependent on inputs from the biosphere, this study also consider damage to ecosystem quality as an complementary assessment at endpoint level (Huijbregts et al., 2016). SimaPro 9 was used to model the system and ReCiPe 2016 was used to assess impact at midpoint and endpoint, which ensured compatibility with established LCIA methods.

3.4. Sensitivity analysis

Four sensitivity analyses were performed to identify the influence of uncertainties on the results, focusing on future energy development, optimised VFA production, increased energy demand and biotic resources. By changing one parameter at a time, the uncertainty related to data and assumption could be quantified. The fist sensitivity analysis reflected the estimated future energy development, aiming to reduce the dependency on fossil fuels and reach the long-term goal of 100% renewable energy (Bosell et al., 2017). Renewables including wind and solar power are projected to dominate future power generation (Newell et al., 2020). If electricity and heat produced from biomass primarily replaced renewable energy, some benefits attributed to biogas might shift. Therefore, future energy development was simulated by substituting the energy included via system expansion with a potential future German energy mix (Table B.1 in Appendix B). According to the United Nations (2019), an optimised production process can promote sustainable use of resources and enhance energy efficiency. The second sensitivity analysis was based on the VFA efficiency assumptions during DF of food waste. Previous research suggests that DF can be optimised to produce up to 25 g VFA/L household food waste, equalling about 5% VFA per unit food waste (Herrero Garcia et al., 2018; Strazzera et al., 2018). Optimised VFA production was simulated by recalculating input and output data in $Energy_{DF}$ assuming that 5 g VFA/kg FW could be extracted. The increased efficiency meant that 84 ton pre-treated food waste was required to produce 4.2 ton VFA, which was also accounted for in $Energy_{AD}$.

Another important aspect to consider is the energy inputs in the *Algae scenario*, especially for algae oil production. This technique is still under development, thus making assumptions regarding the actual energy demand in a large-scale production uncertain. To quantify this uncertainty, all electricity and heat inputs in Table 1, alongside electricity required for VFA separation in Table 2, was increased with 20% to simulate a more energy intensive algae process. The final sensitivity analysis considered the uncertainty of biotic resource depletion in LCA. One aspect of the biodiversity impact is biomass removal from aquatic ecosystems, which affects ecosystems both in terms of resource impact due to the altered *stock level* of the species and impacts on *life support functions*. In a study on sea use impact, Langlois et al. (2015) developed CF for impact on life support functions considering the amount of removed biomass of a certain species. A life support function scenario was simulated using the suggested CF for life support function, where the mass of wild algae removed from ocean in Table 1 was multiplied by 1.4 and the mass of wild anchovy removed from ocean in Table 3 by 13.4.

3.5. Product LCA

Scenario analysis is a valuable tool for evaluating commercial and conceptual system set-ups, while also enabling comparison between the scenarios. However, to increase the applicability of the assessed results outside the modelled scenarios, a product LCA perspective can be applied. Product LCA considers an impact related to a specific product, which facilitates comparison with similar products and in turn can increase the benchmarking properties of the result. The product LCA approach used in this study was conducted by subtracting the environmental impact for $Energy_{AD}$ in both scenarios. Thereby, only the net environmental impact for producing VFA and *algae oil* instead of *fish oil* and energy was assessed. The product LCA assessed the climate impact (kg CO_2eq) per ton DHA and per kg oil, to enable comparison with similar commercial products. In recent studies, the potential to replace DHA in fish feed with DHA from different vegetable sources has been investigated. The climate impact of DHA from the *Algae scenario* and the *Fish scenario* can be compared with similar products, such as Canola or linseed oil. The assessed impact can then be evaluated in comparison with Canola oil and linseed oil, two vegetable oils with high DHA potential that could be used to replace DHA in fish feed (Bélanger-Lamonde et al., 2018; Petrie et al., 2020; Zhou et al., 2019). A 13% bioavailable DHA content was assumed for Canola oil (Petrie et al., 2020), while an ALA content of 56% was assumed for linseed oil (Burns-Whitmore et al., 2019; Karapanagiotidis et al., 2007). Assuming 1% conversion of ALA to DHA in fish, the potential DHA in linseed oil was calculated to be 0.6%. A maximum climate impact for canola rapeseed oil was set to 2.5 kg CO_2eq per L oil (Röös, 2012). Assuming 0.75 kg CO_2eq per kg linseed (Nemecek et al., 2012) and 40% oil content, the maximum climate impact for linseed oil was calculated to be 1.9 kg CO_2eq per kg oil.

4. Results

4.1. Environmental impact

The results showed that for every ton DHA produced in the *Algae scenario*, 14 of 18 midpoint impacts assessed had lower environmental impact compared to the *Fish scenario*, including global warming, terrestrial acidification, and land use (see Table C.1 in Appendix C). As illustrated in Table 5, both scenarios had negative values for global warming and freshwater eutrophication, which means that for every ton DHA produced, the environmental impact was mitigated.

Inclusion of by-products via system expansion was the main reason for the mitigated environmental impact in both the *Algae scenario* (Fig. 4) and the *Fish scenario* (Fig. 5). For global warming and freshwater eutrophication, the highest environmental impact in both scenarios occurred during the pre-treatment of food waste, while the most contributing process for terrestrial acidification and land use was energy production and anaerobic digestion, respectively. *Algae oil* processing caused only about 5% of the total global warming impact for the *Algae scenario*, 10% of land use impact, 4% of terrestrial acidification and 6% of freshwater eutrophication impact. The *fish oil* process caused about 22% of the global warming and land use impact, while about 12% of terrestrial acidification

Table 5
Environmental impact per 1 ton DHA.

	Global warming kg CO_2eq	Terrestrial acidification kg SO_2eq	Freshwater eutrophication kg Peq	Land use m^2a crop eq
Algae scenario	-5.2×10^4	3.5×10^3	-9.4×10^1	2.7×10^3
Fish scenario	-1.5×10^4	3.9×10^3	-9.7×10^1	3.2×10^3

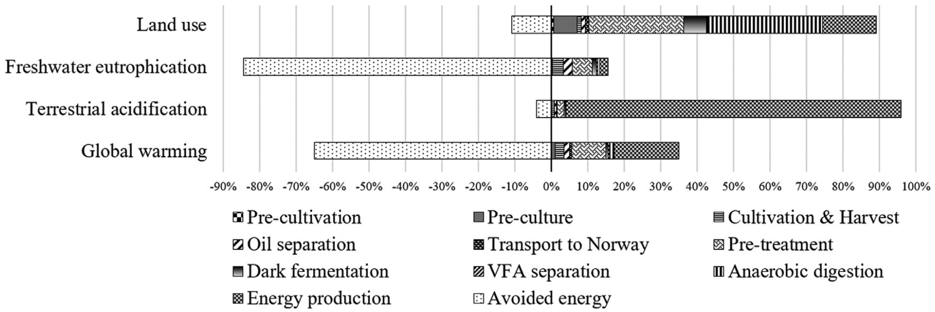

Fig. 4. Environmental impact of the *Algae scenario*, illustrating the percentage of total impact for each production process

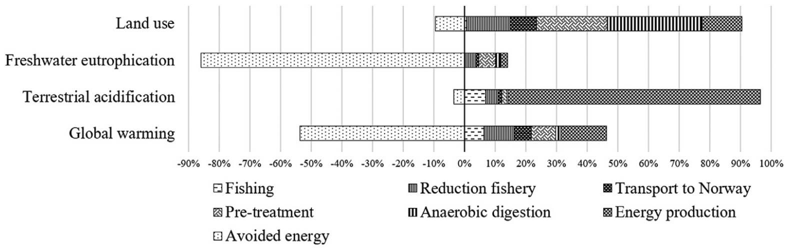

Fig. 5. Environmental impact of the *Fish scenario*, illustraing the percentage of total impact for each production process.

and 5% of freshwater eutrophication originated from it. The fishing process contributed less than 1% of the total land use impact in the *Fish scenario* (Fig. 5).

4.2. Ecosystem damage

The results showed a lower impact for the *Algae scenario* than the *Fish scenario* for nine of 12 endpoint categories assessed, including global warming, terrestrial acidification and land use (see Table C.2 in Appendix C). Total Ecosystem damage per ton DHA produced in the *Algae scenario* was 5.5×10^{-4} species.yr, the *Fish scenario* inferred 8.1×10^{-4} species.yr. The results also showed that *algae oil* and *fish oil* caused Ecosystem damage of 5.1×10^{-5} and 3.1×10^{-4} species.yr, respectively, while the value for both Energy_{DF} and Energy_{AD} was 5.0×10^{-4} species.yr. Similar to the results at midpoint level, *algae oil* processing caused about 10% of the Ecosystem damage, while *fish oil* process caused about 18% (Fig. 6). This suggests that inclusion of by-products had a considerable effect on the total impact on ecosystem quality and effect on biodiversity. As the results illustrated in Fig. 7 illustrate, the largest impact at endpoint level for both scenarios was terrestrial acidification, global warming, freshwater eutrophication and land use. A negative value for Ecosystem damage indicates mitigation of disappeared species per year, whereas a higher positive value can be interpreted as less favourable for biodiversity.

4.3. Sensitivity analysis and product LCA

The results from the sensitivity analysis at midpoint level showed that the *Algae scenario* inferred lower terrestrial acidification impact in comparison to *Fish scenario* even when future energy development, optimised VFA production, increased energy demand and impact on life support function was simulated (Table 6). The *Algae scenario* inferred lower global warming potential for future energy and life support function, but higher global warming potential than the *Fish scenario* for optimised VFA and increased energy. Land use at midpoint was also higher for *Algae scenario* when life support function was simulated.

Even though the global warming potential and land use impact respectively increased for the *Algae scenario* when increased energy and life support function was simulated, the *Fish scenario* still inferred a higher damage to ecosystem quality for all sensitivity analyses preformed (Fig. 8). Ecosystem damage for the *Algae* and *Fish scenarios* increased when future energy production was simulated and decreased when optimised VFA content was assumed. When effects on biotic resources were included via life support functions, the impact for the *Fish scenario* increased markedly, while the Ecosystem damage for the *Algae scenario* remained similar to that in the base case.

The product LCA results showed that the climate impact per ton oil was higher for algae oil than for Canola and linseed oil, but lower than for fish oil (Table 7). Algae oil had the lowest impact

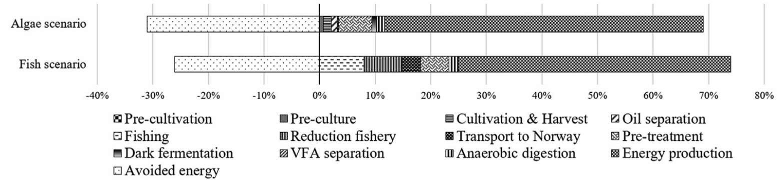

Fig. 6. Ecosystem damage for the *Algae scenario* and *Fish scenario*, illustrating the percentage of total impact for each process.

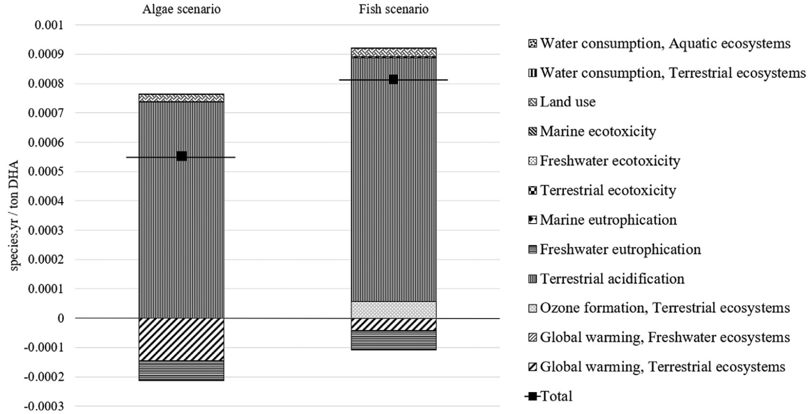

Fig. 7. Ecosystem damage in (left) the Algae scenario and (right) the Fish scenario, illustrating the contribution from each endpoint indicator.

Table 6
Sensitivity analysis result at midpoint level, expressed per 1 ton DHA.

Algae scenario	Global warming kg CO_2eq	Terrestrial acidification kg SO_2eq	Freshwater eutrophication kg Peq	Land use m^2a crop eq
Future energy	2.9×10^4	3.1×10^3	-2.0×10^{-5}	-6.3×10^{-5}
Optimised VFA	-1.0×10^4	1.3×10^3	-3.0×10^1	1.4×10^3
Increased energy	3.0×10^4	3.1×10^3	-9.8×10^1	1.2×10^3
Life support func.	-5.2×10^4	3.5×10^3	-9.4×10^1	2.7×10^3
Fish scenario	Global warming kg CO_2eq	Terrestrial acidification kg SO_2eq	Freshwater eutrophication kg Peq	Land use m^2a crop eq
Future energy	6.7×10^4	3.6×10^3	-2.1×10^{-5}	-4.8×10^{-5}
Optimised VFA	-1.5×10^4	1.7×10^3	-3.1×10^1	1.8×10^3
Increased energy	-1.5×10^4	3.9×10^3	-9.7×10^1	3.2×10^3
Life support func.	2.3×10^5	7.2×10^3	-7.2×10^1	1.8×10^3

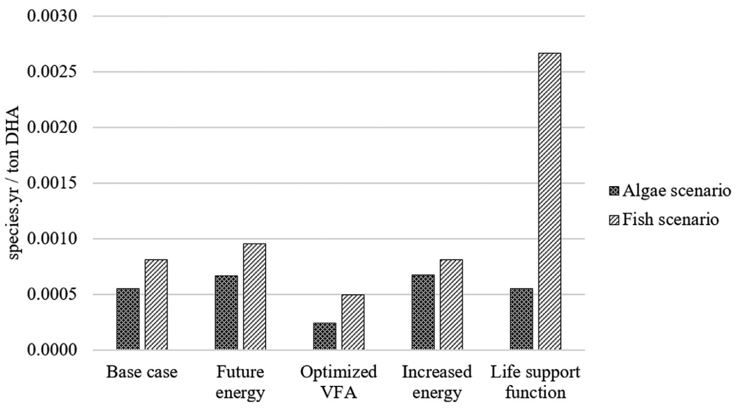

Fig. 8. Results of sensitivity analysis for Ecosystem damage with respect to the Algae scenario and the Fish scenario.

Table 7
Climate impact (product perspective), expressed per ton DHA and per kg oil.

	ton CO_2eq/ton DHA	kg CO_2eq/kg oil
Algae oil[1]	7.6	3.0
Fish oil[1]	44	4.4
Canola oil[2]	23	2.3
Linseed oil[2]	330	1.9

[1]Bioavailable DHA
[2]ALA converted to DHA

per ton of DHA produced, while linseed oil and fish oil had the highest impact in this regard.

5. Discussion

5.1. Environmental impact of DHA from algae oil and fish oil

One of the most important findings in this study was that the environmental impact per ton DHA produced was lower in the *Algae scenario* than in the *Fish scenario* with respect to global warming, terrestrial acidification, land use and Ecosystem damage (Table 5 and Fig. 7). In addition, 20 MWh more electricity and heat was produced in the *Algae scenario*. This, together with overall lower emissions from *algae oil* production compared with *fish oil* production, was the main reason for the lower environmental impact. *Algae oil* production was also less dependent on fossil fuels and electricity, which could explain the lower impact for global warming, terrestrial acidification and eutrophication. Use of a constant CF from midpoint to endpoint likely explains why the climate impact, terrestrial acidification and freshwater eutrophication also contributed most to the impact at endpoint level. As illustrated in Table 5, the impact on water consumption and terrestrial ozone formation was more than 5-fold and 20-fold higher, respectively, in the *Fish scenario*. However, since water consumption and terrestrial ozone formation made a low contribution to the endpoint level, their impact was low when comparing products. The climate impact for *fish oil* was about twice that reported by Silva et al. (2018), owing to a higher contribution from fishing and reduction fishery processing in this study. The difference in climate impact might be explained by different electricity inputs or the updated Ecoinvent datasets, where buildings and machinery are included, which were used to assess impacts in this study. Moreover, most transport in this study was assumed to be of Euro class IV, which reduced the climate impact from transport. The dependency on fossil fuels used for fishing and reduction fishery processing has previously been identified as a contributing factor for the environmental impact of fish oil (Avadí and Fréon, 2013; Cashion et al., 2017; Pelletier and Tyedmers, 2007). Therefore, the impact determined for *fish oil* can be considered consistent with previous findings.

When evaluating the result from this study with previous research it is important to emphasise that all LCA results are highly dependent on given assumptions and methodological choices. Comparing numerical results should thus be done with this in mind, especially regarding impact for the *Algae scenario* since less data and previous LCA research has covered this topic. However, some previous studies have included similar processes as assessed in this study. In a study by Deprá et al. (2020), the environmental impact of multiple algae species, including *C. cohnii* cultivation using glucose as primary carbon feedstock, was assessed. They used a similar method and process for algae cultivation and oil separation as assumed in this study, while also including the impact categories global warming, acidification potential, eutrophication potential and land use. Their result showed an environmental impact of about 178 ton CO_2eq, 0.9 ton SO_2eq, 0.3 ton PO_4eq and 900 m^2 per ton DHA produced. If excluding the contribution from avoided electricity and heat in the *Algae scenario* (Table C.1 in Appendix C), the values for global warming, acidification, eutrophication and land use correspond to 60 ton CO_2eq, 3.6 ton SO_2eq, 0.02 ton PO_4eq and 3000 m^2 per ton DHA. The main difference between the results presented by Deprá et al. (2020) and the results in this study is primarily with respect to global warming and land use. The higher value for global warming is likely caused by the different inputs, such as glucose as primary carbon feedstock and global average instead of the current German electricity mix, while the higher land use is likely a consequence of inclusion of buildings in this study. Even though the numerical results are not perfectly comparable due to different assumptions, the method and overall results can be considered supportive of each other.

Similarly, Schade et al. (2020) conducted a comparative LCA study on fatty acids and protein from microalgae and fish, also including the same midpoint indicators as this study. Several algae species were studied for their potential to accumulate DHA and EPA, including the algae Phaeodactylum tricornutum. By calculating an average impact value from their published supplementary material, result shows an impact of about 2.4 kg CO_2eq, 7.2 g SO_2eq, 8.8 g PO_4eq and 0.17 m^2 per kg dry algae biomass. If assuming a 4% DHA content in the dry biomass from Phaeodactylum tricornutum, the resulting midpoint impact with respect to climate impact, acidification, eutrophication and land use was roughly 60 ton CO_2eq, 0.18 ton SO_2eq, 0.22 ton PO_4eq and 4300 m^2 per ton DHA respectively. The result suggested by Schade et al. (2020) are very close to the environmental impact assessed at midpoint level in this study, when excluding the contribution from the avoided electricity and heat in the *Algae scenario*, even though a different microalgae species and thereby cultivation method was used. To our knowledge, no previous study on algae cultivation has included endpoint indicators for ecosystem quality. Therefore, a comparison with previous studies was not possible even though this aspect is highly requested. The results from this study can thus serve as a basis for future comparisons and enable an improved availability in the emerging field of industrial DHA and EPA production with microalgae. By inclusion of biotic resources, this study also provides a basis for a more holistic future development analysis, where both midpoint and endpoint indications are included in the LCA assessment.

5.2. Environmental impact of energy production and product LCA

Energy production, and the corresponding avoided environmental burden from using the biohydrogen and biogas produced to substitute the German electricity mix, had a considerably high influence on both midpoint and endpoint indicators for the *Algae* and *Fish scenario* (Figs. 4–6). A strong influence of including by-products was also identified in a study by Elginoz et al. (2020), who saw a similar trend when assessing innovative food waste management systems where VFA and methanol were produced. Our results suggested that energy processing was the main contributing factor to terrestrial acidification, while avoided use of lignite for electricity production was the main cause of mitigated freshwater eutrophication, at both midpoint and endpoint level. The main cause of terrestrial acidification during energy production was likely emissions of NOx, NH_3 or SO_2 to air (Huijbregts et al., 2016; Whiting and Azapagic, 2014), caused by the dataset used to describe CHP process of biogas. The avoided use of lignite as an electricity source likely mitigated freshwater eutrophication by reducing phosphorus and nitrogen emissions to soil, air and water (Wang et al., 2015). This is likely the cause for the negative freshwater eutrophication (Table 5). Even though the high influence of terrestrial acidification and freshwater eutrophication on the outcome was unexpected, the same dataset was used in both $Energy_{DF}$ and $Energy_{AD}$, and thereby the same uncertainty applied to both scenarios.

The results for product LCA, where only the net environmental burden of avoided energy was assigned to the *Algae scenario*, showed that algae oil had a lower climate impact per ton DHA produced in comparison to fish oil, canola oil and linseed oil (Table 7). It is important to note, however, that both algae oil and fish oil contain bioavailable DHA, while Canola and linseed oil require conversion of ALA to provide DHA. Conversion rate of ALA to DHA can vary greatly, so the actual DHA per ton oil for canola oil and linseed oil shows a large range, meaning that direct comparisons with algae oil and fish oil should be performed with caution. However, both canola oil and algae oil have established potential to replace fish oil in fish feed (Bélanger-Lamonde et al., 2018; Cottrell et al., 2020; Petrie et al., 2020).

5.3. Uncertainties

According to Hetherington et al. (2014), the parameter uncertainties are often higher for processes using emerging technologies and early-stage LCAs, where production, inputs and outputs have not yet been fully established. The *Algae scenario* can be considered an emerging technology, for which the accessibility of inventory data was the main source of uncertainty. For instance, Ecoinvent 3.5 contains data on fishing activities and Peruvian site locations (Avadí et al., 2020), but datasets for algae aquaculture, DF and VFA separation are currently not available. Similar uncertainty was associated with the results for Ecosystem damage and potential loss of biodiversity, where the numerical results, especially for the *Fish scenario*, were likely strongly underestimated. Although, the removal of primary producers could cause damage to higher tropic levels, which should be considered in future assessments. Moreover, the LCI for Peruvian anchovy represents sustainable fishing, even though unsustainable fishing is an established problem (Fréon et al., 2017). If the anchovy used for fish oil were sourced from unsustainable fishing, the Ecosystem damage for the *Fish scenario* would be higher than reported in this study. A final uncertainty aspect was the representation of species.yr in the ReCiPe endpoint approach, where the same weight is assigned to all species of plants and lower organisms, thus not accounting for endangered or overexploited species. Due to the uncertainty, the impact on biodiversity at endpoint level should primarily be used to identify hotspots for actions to reduce environmental burden.

The results from the sensitivity analysis suggested that AD of food waste to produce biogas for electricity and heat can be less favourable as a valorisation method including the material use in the future (Fig. 8), since the energy can no longer replace fossil energy sources. This is an important finding for future developments within food waste management, especially since the method for re-use should be consistent with the most efficient option to optimise resource recovery (Teigiserova et al., 2020). Another important consideration is that the results show the impact primarily using German energy mix and Euro IV road transport, which provide a site-specific result and limited transport emissions. The decreased environmental impact when less food waste was required to produce VFA was likely mainly due to the reduced amount of energy and building required for processing. Based on previous research (Barr and Landis, 2018; Keller et al., 2017; Taelman et al., 2013), the *Algae scenario* could be improved by nutrient recycling (e.g. recycled algae culture medium or re-using additional nutrients from VFA), by efficient energy use (e.g. using methane as fuel for transportation) or by upscaling the production process. In comparison, the fishing and reduction fishery process has been optimised and streamlined for decades, and therefore does not offer the same development potential. Since algae cultivation can be sensitive to trace components, there is a considerable uncertainty related to the inputs and outputs for this process. As the sensitivity result for optimised VFA production and increased energy show (Table 6), the global warming potential increase for *Algae scenario*, while the land use impact increase when life support function was accounted for. The result indicates that depending on the required energy input the result is uncertain. This highlights the importance of further research and development within algae cultivation for oil accumulation to further increase the knowledge and data availability.

The *Fish scenario* resulted in over 3-fold higher Ecosystem damage in comparison with the base case when the life support function was included. Since more biomass of Peruvian anchovy than wild-type *C. cohnii* was required per ton DHA and since anchovy had a higher trophic level, the CF was higher for anchovy. Direct impact translation to species.yr is not yet possible using available LCIA methods, but this result provides a crucial indication of the magnitude of sensitivity related to Ecosystem damage and effects on biodiversity within LCA. The impact of biotic resources have been identified in previous studies (Avadí and Fréon, 2013; Winter et al., 2017; Woods et al., 2016), where the main conclusion was the importance of developing methods that include multiple aspects of biotic resources in LCA. The results from the present study confirm this conclusion. Another important finding in this study was that even though the *Algae scenario* inferred higher impact at midpoint for multiple sensitivity scenarios, the result for Ecosystem damage at endpoint level was consistently lower in comparison to the *Fish scenario* (Fig. 8). Even though a higher uncertainty is related to impact at endpoint level, alongside the relative uncertainty related to impact on biotic resources in LCA, the results provide an important indication of the environmental relevance which is vital for a more holistic assessment in the future.

5.4. Future outlook

An important aspect of sustainable development is to ensure that an increasing global population has access to nutritiously valuable food (FAO, 2020, 2019). Threats to food security due to increased temperatures and changes in natural ecosystem functions are likely to emerge as tangible consequences of climate change and loss of biodiversity. According to Avadí et al. (2020) and FAO (2018), the aquaculture sector will likely continue to grow, while natural DHA synthesis by marine microalgae is estimated to decrease due to global warming (Colombo et al., 2020). Therefore, developing new production methods within the food supply chain, preferably with increased resource recovery to reduce the environmental impact and damage to ecosystem quality, can be considered the most urgent global challenge of today. In the near future, it will also become increasingly important to develop alternative ways to produce DHA (Beal et al., 2018; Cottrell et al., 2020; Russo et al., 2021).

Algae oil production has the potential to expand and meet multiple demands of the growing aquaculture sector, while also lowering the burden on wild-caught fisheries and decreasing the use of biotic resources (Barr and Landis, 2018; Ghamkhar and Hicks, 2020) in comparison with traditional fish feed (Taelman et al., 2013). Given the globality of the current supply chain, using bioreactors for algae cultivation enables a local primary production of DHA which likely will require a shorter transport distance and opportunities for an increased degree of self-sufficiency. In a global context, this is especially relevant with respect to global food security and energy production. As suggested by Chalima et al. (2019), the development of microalgae cultivation processes for oil accumulation has so far been slow, primarily due to the relatively high economic costs related to the emerging technology for algae cultivation and harvest. At present, algae oil would therefore likely result in higher economic costs than fish oil

(Sprague et al., 2017; Yarnold et al., 2019), since it requires further development to be implemented in large-scale production. However, since sustainable development requires a shift from a linear to a circular bioeconomy, utilising available resources from other production systems will become increasingly important to meet the needs of future generations. DHA, as produced in the *Algae scenario*, can bridge a future gap between supply and demand, while short-chain carboxylic acids produced alongside biohydrogen and biogas could be integrated into existing infrastructure for common food waste valorisation methods quite rapidly. To promote the evolution of future technologies with lower environmental impact that also require fewer natural resources to produce products with maintained quality and nutrient composition, companies and policy makers must overcome the hurdle of uncertainty related to new innovations. This is also concluded by Deprá et al. (2020), who also stress the importance of abandon current technology to enable a transition to more sustainable solutions. Previous studies indicate that promoting circular resource use will become increasingly important in the future (Jovanovic et al., 2021; Oliver et al., 2020; Russo et al., 2021). One could therefore argue that fish oil in fish feed should be replaced with algae oil. Important to consider however, is that alternative production methods should be carefully assessed to prevent fish oil being replaced with another potentially unsustainable DHA source. The results in this study indicate that algae are a more sustainable source of DHA than fish, but other sources of DHA should also be evaluated, for instance single cell oils derived from yeast or genetically modified vegetable sources. This is especially important given the uncertainty related to LCA for technologies and solutions that are still under development.

Given that quantities required in the *Algae scenario* are 1:1 scalable, only about 2.7% of the globally available food waste generated at households, retail or service sector is required to produce enough VFA via DF to substitute the yearly demand of about 100 000 tons of DHA from fish oil in aquafeed. Even though enough food waste is already available, it is important to consider potential burden shifts when replacing fish oil with algae oil, such as an increased eutrophication potential (Table 6). However, this also illustrate the importance of investing in further development of this technology since it has a large potential to increase resource efficiency and promote both sustainable aquaculture and improved resource recovery within food waste management. Technologies that favour circular flows, with resource recovery and nutrient recycling, could also contribute to several SDGs (Herrero et al., 2020; Teigiserova et al., 2020). To fully assess the impact on SDG fulfilment, new LCA frameworks are currently being developed (Life Cycle Initiative, 2021; Weidema et al., 2020). Although, the results in this study indicate that, compared with fish oil, DHA from the *Algae scenario* could contribute to multiple SDGs. For instance *climate action* by reduced greenhouse gas emissions (SDG 13), *responsible consumption and production* by increased resource efficiency and reduced waste (SDG 12), *zero hunger* by supporting sustainable food production systems (SDG 2), *affordable and clean energy* by increasing the share of renewable energy in the global energy mix (SDG 7), and ultimately mitigate loss of biodiversity and damage to ecosystem quality by reducing acidification, eutrophication and habitat degradation (SDGs 14 and 15). Given the estimated expansion of aquaculture and increased global demand for food rich in DHA, algae-based aquafeeds thus represent an alternative production method using a carbon source derived from already available resources.

5.5. Recommendations for future studies

As established by previous studies, these is an urgent need for life cycle assessments and more data to enable environmental assessments and sustainability evaluation of emerging food and aquaculture technologies, including microalgae cultivation (Avadí et al., 2020; Lopes da Silva et al., 2019). In order to mimic the composition of fish oil better in biotechnologically derived cell suspensions, algae species, e.g. *Schizochytrium* sp., that produce EPA and DHA in a ratio similar to that in fish oil, could be considered (Hart et al., 2021; Sprague et al., 2015). Alternatively, DHA from *C. cohnii* could potentially replace fish oil supplements in human diets. Some hurdles need to be overcome, however. First, bioproduction needs to be further optimised and made robust, e.g. through integration of suitable online monitoring that allows variable utilisation of feedstock while maintaining cell viability and production capacity, e.g. through technologies that capture the single cells (Delvigne et al., 2018). Under current EU animal by-product legislation, DHA from the *Algae scenario* can only be implemented if exclusively vegetable waste is used to produce VFA, which might have an impact on the yield of DF (Strazzera et al., 2018). This requires further investigation and optimisation of such processes based on cell physiology, as only cells with a certain metabolic turnover excrete VFAs (Bockisch et al., 2018). The best option for food waste valorisation should also be examined in more detail, including both small- and large-scale case studies. This is especially important since we are still a long way from a harmonized assessment of food waste management, which makes it difficult to fully assess the future potential for suggested solutions.

When comparing certain production scenarios, direct effects on biodiversity cannot be assessed with established LCIA methods, so future studies would benefit from including this aspect. Since resource depletion, overfishing and invasive species are some of the main threats to loss of marine biodiversity (Woods et al., 2016), these aspects should be included in studies with a marine biodiversity focus. This could potentially be accomplished by implementing CFs for biotic resource depletion (Hélias et al., 2018), overfishing (Emanuelsson et al., 2014) or invasive species (Hanafiah et al., 2013). To support sustainable development and maintain a rich biodiversity, there is an urgent need for robust and extensive impact assessment methods to account for the full impact on biotic resources (Avadí et al., 2020; Asselin et al. 2020; Winter et al., 2017). One could therefore argue that the most important future research needs are to reduce knowledge gaps and to develop LCA methods that cover all five drivers of biodiversity loss.

6. Conclusions

This study assessed the environmental impact of a conceptual *Algae scenario* with DHA produced via the microalgae *C. cohnii* grown in a bioreactor cultivation process using VFA extracted from food waste as its main carbon feedstock. The utilization of VFA from dark fermentation enables a combination of bioconversion for a high added-value DHA product via microalgae with renewable energy production. The impact was compared with that of a conventional *Fish scenario* with DHA derived from Peruvian anchovy. Alongside the environmental impact at midpoint level, the important aspect of ecosystem quality at endpoint level was also assessed using Ecosystem damage as indicator for biodiversity loss. The main by-product in both scenarios was electricity and heat, included via system expansion. The global warming, terrestrial acidification, freshwater eutrophication and land use per ton DHA produced in the *Algae scenario* was found to be -52 ton CO_2eq, 3.5 ton SO_2eq, -94 kg Peq, 2700 m^2 eq, respectively. In comparison, the impact per ton DHA in the *Fish scenario* was -15 ton CO_2eq, 3.9 ton SO_2eq, -97 kg Peq and 3200 m^2 eq. The Ecosystem damage for Algae scenario and Fish scenario was 5.5×10^{-4} and 8.1×10^{-4} species per year respectively. Even though established LCIA methods only assess indirect effects on biodiversity, the *Algae scenario* resulted in lower Ecosystem damage than the *Fish scenario* even when a future energy development, optimized VFA production, increased energy demand and effects on biotic resources were considered via sensitivity analyses. This study also showed that included by-products and energy production had a high influence on the total environmental burden for both scenarios. From a product LCA perspective, algae oil had the lowest climate impact per ton DHA for all evaluated oils.

At present, coupling DF and subsequent monocultivation for DHA and EPA production is still under development and thus require higher economic investments to enable a large-scale production similar to traditional fish oil processing. However, the production of valuable DHA with lower environmental impact arguably justifies a higher production cost especially since it also provides an improved food waste valorisation solution and a source of renewable energy. The environmental aspect must be considered a key component in both political decision making and company development to fully achieve a sustainable development, as well as enabling a shift from a linear to a circular bioeconomy where available resources are recovered and used in the most efficient way possible. The result from this study emphasise that Algae oil holds a promising potential to increase sustainability within aquaculture, provided that continued development and optimization of the technology and process is enabled through active decision-making, purposeful investments, and further research.

This study showed that DHA produced by microalgae using VFA from DF of food waste can reduce loss of biodiversity and support sustainable production while satisfying increased future demand for DHA within the food supply chain. This could support sustainable development by meeting current needs for DHA without compromising nature's ability to produce this essential fatty acid in the future. The *Algae scenario* approach also enabled increased resource efficiency by recovering nutrients and resources in food waste for value addition. By using agricultural and food industry by-products to produce DHA, overfishing, for example of Peruvian anchovy, could be counteracted and thereby increasing the overall sustainability of aquaculture while maintaining essential ecosystem quality.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

The study was conducted as part of the project 'Adding value in resource effective food systems' (AVARE) funded by the H2020 ERA-net Cofund on Sustainable Food Production and Consumption (SUSFOOD2) by the Swedish Research Council for Sustainable Development (Formas), grant number FR-2018/0001, the Finnish Ministry of Agriculture and Forestry grant number 56/03.01.02/2018, and the German Federal Ministry of Food and Agriculture (BMEL) administered by the German Federal Office for Agriculture and Food (BLE), grant number 2818ERA07B.

Appendix A

Ecoinvent datasets used in SimaPro 9 (PhD licence) to model impact for the *Algae scenario* and the *Fish scenario*. Original data generator, dataset name, geographical location and process modelled using the dataset are shown. All datasets were taken from Ecoinvent 3.5 with the allocation cut-off by classification setting.

Table A.1
Ecoinvent datasets used in SimaPro, using the allocation cut-off by classification setting.

Dataset number	Data generator	Dataset name	Location	Used to model:
1	Symeonidis, A.	*Market for marine fish*	GLO	Wild *C. cohnii*
2	System	*Market for nitrogen fertiliser, as N*	GLO	N fertiliser
3	System	*Market for phosphate fertiliser, as P2O5*	GLO	P fertiliser
4	System	*Market for glucose*	GLO	Glucose
5	System	*Market group for tap water*	RER	Water
6	Faist E., M.	*Market for natural gas, high pressure*	DE	Natural gas
7	System	*Market for steam, in chemical industry*	GLO	Steam
8	Dux, D.	*Liquid manure storage and processing facility construction*	CH	Algae processing
9	Valsasina, L.	*Market for transport, freight, lorry 3.5-7.5 metric ton, EURO4*	RER	Transport
10	PRé Consultants	*Carbon dioxide, unspecified*	n.a.	CO_2 to air
11	System	*Market for fodder yeast*	GLO	Yeast
12	System	*Market for sodium chloride, brine solution*	GLO	Reef salt
13	System	*Market for molasses, from sugar beet*	GLO	Molasses
14	Valsasina, L.	*Market for transport, freight, lorry 7.5-16 metric ton, EURO4*	RER	Transport
15	System	*Market for wastewater, average*	Europe without Switzerland	Wastewater
16	Gnansounou, E.	*Oil mill construction*	CH	Oil separation
17	Kägi, T.	*Waste preparation facility construction*	CH	FW processing
18	Doka, G.	*Market for municipal waste collection service, by 21 metric ton lorry*	CH	Transport
19	PRé Consultants	*Recycling of aluminium*	GLO	Metal reject
20	Treyer, K.	*Treatment of municipal solid waste, incineration*	DE	Plastic reject
21	Schleiss, K.	*Anaerobic digestion plant construction, agriculture, with methane recovery*	CH	DF and AD

(*continued on next page*)

Table A.1 (*continued*)

Dataset number	Data generator	Dataset name	Location	Used to model:
22	System.	*Market for vegetable oil refinery*	GLO	VFA processing
23	Treyer, K. Ruiz, EM.	*Heat and power co-generation, biogas, gas engine Excluding contribution from: market for biogas*	DE RoW	CHP plant
24	Treyer, K.	*Heat and power co-generation, natural gas, combined cycle power plant, 400MWelectrical*	DE	Heat (fossil)
25	Avadi, A.	*Anchovy, capture by steel purse seiner and landing whole, fresh*	PE	Fishing anchovy
26	Avadi, A.	*Fishmeal and fish oil production, 63-65% protein*	PE	Reduction fishery
27	Simons, A.	*Transport, freight, lorry 7.5-16 metric ton, EURO4*	RoW	Transport
28	System	*Market for transport, freight, sea, transoceanic ship with reefer, cooling*	GLO	Transport

Appendix B

In 2020, about 50% of German electricity was generated from renewable sources (Energy-Charts, 2020), while energy use for heat was dominated by fossil fuels (Euroheat & Power, 2019; IEA, 2020). To accurately represent the use of electricity, a custom energy mix was created in SimaPro 9 using Ecoinvent 3.5 datasets. Current electricity mix represent the 2020 production mix, while the future electricity mix consist of entirely renewable energy. The fossil energy dataset is used to assess the avoided energy in Algae scenario and Fish scenario, that is replaced with electricity and heat from EnergyDF and EnergyAD respectively.

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Table B.1
Datasets representing the share of energy source used to produce current, fossil, and future energy mix. Net electricity distribution was sourced from Energy-Charts (2020) on 18 December 2020.

	Data generator	Dataset name	Location	%	Used to model
(1) **Current electricity mix**					
	Treyer, K.	*Electricity production, lignite*	DE	16.9	Electricity mix
	Treyer, K.	*Electricity production, nuclear, pressure water reactor*	DE	12.4	Electricity mix
	System	*Electricity production, natural gas, 10MW*	DE	12.1	Electricity mix
	Treyer, K.	*Electricity production, wind, >3MW turbine, onshore*	DE	26.6	Electricity mix
	Treyer, K.	*Electricity production, photovoltaic, 570kWp open ground installation, multi-Si*	DE	10.7	Electricity mix
	Treyer, K.	*Heat and power co-generation, biogas, gas engine*	DE	9.4	Electricity mix
	Treyer, K.	*Electricity production, hard coal*	DE	7.3	Electricity mix
	Treyer, K.	*Electricity production, hydro, pumped storage*	DE	3.9	Electricity mix
	System	*Electricity, high voltage, production mix*	DE	0.7	Electricity mix
(1) **Fossil electricity mix**					
	Treyer, K.	*Electricity production, lignite*	DE	33	Electricity mix
	System	*Electricity production, natural gas, 10MW*	DE	33	Electricity mix
	Treyer, K.	*Electricity production, hard coal*	DE	33	Electricity mix
(1) **Future electricity mix**					
	Treyer, K.	*Electricity production, wind, >3MW turbine, onshore*	DE	45	Electricity mix
	Treyer, K.	*Electricity production, photovoltaic, 570kWp open ground installation, multi-Si*	DE	45	Electricity mix
	Treyer, K.	*Heat and power co-generation, biogas, gas engine*	DE	10	Electricity mix

Appendix C

Table C.1

Environmental impact for each process in the *Algae scenario* and the *Fish scenario*, expressed per 1 ton DHA. GWP = Global warming [kg CO2 eq], SOD = Stratospheric ozone depletion [kg CFC11 eq], IR = Ionizing radiation [kBq Co-60 eq], OF_{HH} = Ozone formation Human health [kg NOx eq], FPMF = Fine particulate matter formation [kg PM2.5 eq], OF_{TE} = Ozone formation - Terrestrial ecosystems [kg NOx eq], TA = Terrestrial acidification [kg SO2 eq], FE = Freshwater eutrophication [kg Peq], ME = Marine eutrophication [kg N eq], TETox = Terrestrial ecotoxicity [kg 1,4-DCB], FETox = Freshwater ecotoxicity [kg 1,4-DCB], METox = Marine ecotoxicity [kg 1,4-DCB], HCTox = Human carcinogenic toxicity [kg 1,4-DCB], HNCTox = Human non-carcinogenic toxicity [kg 1,4-DCB], LU = Land use [m2a crop eq], MRS = Mineral resource scarcity [kg Cu eq], FRS = Fossil resource scarcity [kg oil eq], WC = Water consumption [m3].

	GWP	SOD	IR	OF_{HH}	FPMF	OF_{TE}	TA	FE	ME	TETox	FETox	METox	HCTox	HNCTox	LU	MRS	FRS	WC
Algae oil																		
Pre-cultivation	3.4×10^{1}	1.5×10^{-4}	2.1×10^{0}	8.1×10^{-2}	5.4×10^{-2}	8.2×10^{-2}	1.8×10^{-1}	1.4×10^{-2}	4.1×10^{-2}	1.1×10^{2}	1.0×10^{0}	1.5×10^{0}	1.1×10^{0}	3.9×10^{1}	2.0×10^{1}	1.0×10^{-1}	7.7×10^{0}	9.8×10^{-1}
Pre-culture	1.6×10^{3}	2.2×10^{-3}	4.7×10^{1}	5.7×10^{0}	2.2×10^{0}	5.8×10^{0}	5.7×10^{0}	3.0×10^{-1}	3.2×10^{-1}	1.5×10^{4}	4.4×10^{1}	6.7×10^{1}	5.0×10^{1}	1.4×10^{3}	2.2×10^{2}	5.5×10^{0}	4.7×10^{2}	3.3×10^{1}
Cultivation & harvest	4.2×10^{3}	2.8×10^{-3}	5.9×10^{2}	4.0×10^{0}	4.4×10^{0}	4.0×10^{0}	2.7×10^{1}	4.1×10^{0}	3.7×10^{-1}	4.1×10^{3}	2.1×10^{2}	2.7×10^{2}	2.1×10^{2}	3.5×10^{3}	4.4×10^{1}	4.4×10^{0}	8.0×10^{2}	-4.4×10^{0}
Oil separation	2.6×10^{3}	2.3×10^{-3}	4.9×10^{2}	3.3×10^{0}	3.7×10^{0}	3.3×10^{0}	2.2×10^{1}	3.4×10^{0}	2.2×10^{-1}	3.5×10^{3}	1.7×10^{2}	2.3×10^{2}	1.8×10^{2}	2.9×10^{3}	3.8×10^{1}	3.8×10^{0}	6.7×10^{2}	9.9×10^{0}
Transport to Norway	9.9×10^{2}	4.3×10^{-4}	1.9×10^{1}	4.1×10^{0}	1.2×10^{0}	4.2×10^{0}	2.9×10^{0}	9.1×10^{-2}	6.9×10^{-3}	1.1×10^{4}	1.8×10^{1}	3.0×10^{1}	2.4×10^{1}	6.7×10^{2}	3.5×10^{1}	2.3×10^{0}	3.4×10^{2}	2.8×10^{0}
$Energy_{DF}$																		
Pre-treatment	1.6×10^{4}	1.2×10^{-2}	9.8×10^{2}	7.3×10^{1}	2.3×10^{1}	7.7×10^{1}	7.5×10^{1}	7.5×10^{0}	5.0×10^{-1}	4.5×10^{4}	1.5×10^{3}	2.0×10^{3}	7.7×10^{2}	3.1×10^{4}	9.1×10^{2}	4.8×10^{1}	4.1×10^{3}	4.5×10^{1}
Dark fermentation	1.2×10^{3}	1.0×10^{-3}	2.1×10^{2}	1.9×10^{0}	1.9×10^{0}	1.9×10^{0}	9.4×10^{0}	1.4×10^{0}	9.1×10^{-2}	3.1×10^{3}	8.1×10^{1}	1.1×10^{2}	1.1×10^{2}	1.6×10^{3}	2.2×10^{2}	5.7×10^{0}	3.0×10^{2}	1.3×10^{2}
VFA separation	4.2×10^{2}	3.7×10^{-4}	7.8×10^{1}	5.4×10^{-1}	6.1×10^{-1}	5.4×10^{-1}	3.6×10^{0}	5.5×10^{-1}	3.5×10^{-2}	6.3×10^{2}	2.8×10^{1}	3.7×10^{1}	2.8×10^{1}	4.8×10^{2}	6.6×10^{0}	7.3×10^{-1}	1.1×10^{2}	1.6×10^{0}
Anaerobic digestion	2.3×10^{3}	1.2×10^{-3}	1.0×10^{2}	8.6×10^{0}	3.8×10^{0}	8.8×10^{0}	8.5×10^{0}	7.7×10^{-1}	4.8×10^{-2}	2.4×10^{4}	1.0×10^{2}	1.5×10^{2}	1.8×10^{2}	3.2×10^{3}	1.1×10^{3}	2.4×10^{1}	7.0×10^{2}	1.6×10^{1}
Energy production	3.0×10^{4}	2.6×10^{-1}	2.6×10^{2}	2.4×10^{1}	4.4×10^{2}	2.4×10^{1}	3.5×10^{3}	3.2×10^{0}	2.0×10^{-1}	1.5×10^{4}	2.1×10^{2}	2.9×10^{2}	3.2×10^{2}	5.6×10^{3}	5.2×10^{2}	2.7×10^{1}	1.6×10^{3}	2.2×10^{1}
Avoided energy	-1.1×10^{5}	-4.2×10^{-2}	-5.2×10^{2}	-1.5×10^{2}	-5.1×10^{1}	-1.5×10^{2}	-1.5×10^{2}	-1.2×10^{2}	-7.1×10^{0}	-1.8×10^{4}	-2.9×10^{3}	-4.0×10^{3}	-5.6×10^{3}	-8.3×10^{4}	-3.8×10^{2}	-2.7×10^{1}	-3.2×10^{4}	-2.0×10^{2}
Fish oil																		
Fishing	1.3×10^{4}	3.2×10^{-3}	1.5×10^{2}	2.8×10^{2}	9.1×10^{1}	2.8×10^{2}	2.9×10^{2}	2.3×10^{-1}	3.0×10^{-2}	7.7×10^{3}	1.8×10^{1}	1.2×10^{3}	5.7×10^{1}	7.0×10^{2}	2.3×10^{1}	3.4×10^{0}	4.3×10^{3}	2.1×10^{1}
Reduction fishery	2.0×10^{4}	4.0×10^{-3}	1.7×10^{2}	1.2×10^{2}	5.9×10^{1}	1.2×10^{2}	1.7×10^{2}	4.8×10^{0}	3.3×10^{-1}	1.8×10^{4}	2.4×10^{2}	7.6×10^{2}	3.9×10^{2}	7.6×10^{3}	5.6×10^{2}	1.0×10^{1}	4.4×10^{3}	1.6×10^{2}
Transport to Norway	1.1×10^{4}	5.5×10^{-3}	2.2×10^{2}	6.6×10^{1}	2.0×10^{1}	6.7×10^{1}	5.0×10^{1}	1.1×10^{0}	8.5×10^{-2}	1.1×10^{5}	2.0×10^{2}	3.3×10^{2}	2.8×10^{2}	6.9×10^{3}	3.4×10^{2}	2.5×10^{1}	3.9×10^{3}	3.1×10^{1}
$Energy_{AD}$																		
Pre-treatment	1.6×10^{4}	1.2×10^{-2}	9.8×10^{2}	7.3×10^{1}	2.3×10^{1}	7.7×10^{1}	7.5×10^{1}	7.5×10^{0}	5.0×10^{-1}	4.5×10^{4}	1.5×10^{3}	2.0×10^{3}	7.7×10^{2}	3.1×10^{4}	9.1×10^{2}	4.8×10^{1}	4.1×10^{3}	4.5×10^{1}
Anaerobic digestion	3.4×10^{3}	2.2×10^{-3}	3.0×10^{2}	1.0×10^{1}	5.4×10^{0}	1.0×10^{1}	1.7×10^{1}	2.1×10^{0}	1.3×10^{-1}	2.6×10^{4}	1.8×10^{2}	2.5×10^{2}	2.7×10^{2}	4.5×10^{3}	1.2×10^{3}	2.7×10^{1}	9.7×10^{2}	1.4×10^{2}
Energy production	3.0×10^{4}	2.6×10^{-1}	2.6×10^{2}	2.4×10^{1}	4.4×10^{2}	2.4×10^{1}	3.5×10^{3}	3.2×10^{0}	2.0×10^{-1}	1.5×10^{4}	2.1×10^{2}	2.9×10^{2}	3.2×10^{2}	5.6×10^{3}	5.2×10^{2}	2.7×10^{1}	1.6×10^{3}	2.2×10^{1}
Avoided energy	-1.1×10^{5}	-4.2×10^{-2}	-5.1×10^{2}	-1.5×10^{2}	-5.0×10^{1}	-1.5×10^{2}	-1.5×10^{2}	-1.2×10^{2}	-7.1×10^{0}	-1.7×10^{4}	-2.9×10^{3}	-4.0×10^{3}	-5.6×10^{3}	-8.3×10^{4}	-3.8×10^{2}	-2.6×10^{1}	-3.1×10^{4}	-2.0×10^{2}

Table C.2
Ecosystem damage [species.yr] for each process in the *Algae scenario* and the *Fish scenario*, expressed per 1 ton DHA. GWP_{TE} = Global warming - Terrestrial ecosystems, GWP_{FE} = Global warming - Freshwater ecosystems, OF_{TE} = Ozone formation - Terrestrial ecosystems, TA = Terrestrial acidification, FE = Freshwater eutrophication, ME = Marine eutrophication, TETox = Terrestrial ecotoxicity, FETox = Freshwater ecotoxicity, METox = Marine ecotoxicity, LU = Land use, WC_{TE} = Water consumption - Terrestrial ecosystems, WC_{AE} = Water consumption - Aquatic ecosystems.

	GWP_{TE}	GWP_{FE}	OF_{TE}	TA	FE	ME	TETox	FETox	METox	LU	WC_{TE}	WC_{AE}
Algae oil												
Pre-cultivation	9.5×10^{-8}	2.6×10^{-12}	1.1×10^{-8}	3.8×10^{-8}	9.2×10^{-9}	7.0×10^{-11}	1.3×10^{-9}	7.1×10^{-10}	1.5×10^{-10}	1.8×10^{-7}	9.7×10^{-9}	4.5×10^{-13}
Starting culture	4.5×10^{-6}	1.2×10^{-10}	7.5×10^{-7}	1.2×10^{-6}	2.0×10^{-7}	5.5×10^{-10}	1.7×10^{-7}	3.0×10^{-8}	7.0×10^{-9}	2.0×10^{-6}	4.0×10^{-7}	2.9×10^{-11}
Cultivation & harvest	1.2×10^{-5}	3.2×10^{-10}	5.2×10^{-7}	5.7×10^{-6}	2.7×10^{-6}	6.2×10^{-10}	4.7×10^{-8}	1.4×10^{-7}	2.9×10^{-8}	3.9×10^{-7}	-9.0×10^{-8}	-3.9×10^{-12}
Oil separation	7.3×10^{-6}	2.0×10^{-10}	4.3×10^{-7}	4.7×10^{-6}	2.3×10^{-6}	3.7×10^{-10}	4.0×10^{-8}	1.2×10^{-7}	2.4×10^{-8}	3.4×10^{-7}	1.1×10^{-7}	4.9×10^{-12}
Transport to Norway	2.8×10^{-6}	7.5×10^{-11}	5.4×10^{-7}	6.1×10^{-7}	6.1×10^{-8}	1.2×10^{-11}	1.2×10^{-7}	1.3×10^{-8}	3.2×10^{-9}	3.1×10^{-7}	3.2×10^{-8}	1.6×10^{-12}
$Energy_{DF}$												
Pre-treatment	4.5×10^{-5}	1.2×10^{-9}	9.9×10^{-6}	1.6×10^{-5}	5.0×10^{-6}	8.5×10^{-10}	5.1×10^{-7}	1.0×10^{-6}	2.1×10^{-7}	8.1×10^{-6}	3.8×10^{-7}	1.8×10^{-11}
Dark fermentation	3.3×10^{-6}	9.1×10^{-11}	2.5×10^{-7}	2.0×10^{-6}	9.5×10^{-7}	1.6×10^{-10}	3.6×10^{-8}	5.6×10^{-8}	1.1×10^{-8}	1.9×10^{-6}	1.7×10^{-6}	7.5×10^{-11}
VFA separation	1.2×10^{-6}	3.2×10^{-11}	7.0×10^{-8}	7.7×10^{-7}	3.7×10^{-7}	6.0×10^{-11}	7.2×10^{-9}	2.0×10^{-8}	3.9×10^{-9}	5.9×10^{-8}	1.7×10^{-8}	8.0×10^{-13}
Anaerobic digestion	6.5×10^{-6}	1.8×10^{-10}	1.1×10^{-6}	1.8×10^{-6}	5.1×10^{-7}	8.1×10^{-11}	2.7×10^{-7}	7.1×10^{-8}	1.6×10^{-8}	9.8×10^{-6}	1.5×10^{-7}	7.6×10^{-12}
Energy production	8.5×10^{-5}	2.3×10^{-9}	3.1×10^{-6}	7.4×10^{-4}	2.1×10^{-6}	3.4×10^{-10}	1.7×10^{-7}	1.5×10^{-7}	3.1×10^{-8}	4.6×10^{-6}	1.9×10^{-7}	1.8×10^{-11}
Avoided energy	-3.1×10^{-4}	-8.5×10^{-9}	-1.9×10^{-5}	-3.2×10^{-5}	-7.7×10^{-5}	-1.2×10^{-8}	-2.0×10^{-7}	-2.0×10^{-6}	-4.2×10^{-7}	-3.4×10^{-6}	-2.5×10^{-6}	-1.1×10^{-10}
Fish oil												
Fishing	3.6×10^{-5}	9.9×10^{-10}	3.7×10^{-5}	6.1×10^{-5}	1.6×10^{-7}	5.1×10^{-11}	8.8×10^{-8}	1.3×10^{-8}	1.3×10^{-7}	2.0×10^{-7}	2.7×10^{-7}	1.3×10^{-11}
Reduction fishery	5.5×10^{-5}	1.5×10^{-9}	1.5×10^{-5}	3.6×10^{-5}	3.2×10^{-6}	5.6×10^{-10}	2.0×10^{-7}	1.7×10^{-7}	8.0×10^{-8}	5.0×10^{-6}	1.9×10^{-6}	1.4×10^{-10}
Transport to Norway	3.2×10^{-5}	8.8×10^{-10}	8.6×10^{-6}	1.1×10^{-5}	7.2×10^{-7}	1.4×10^{-10}	1.2×10^{-6}	1.4×10^{-7}	3.5×10^{-8}	3.1×10^{-6}	3.6×10^{-7}	1.9×10^{-11}
$Energy_{AD}$												
Pre-treatment	4.5×10^{-5}	1.2×10^{-9}	9.9×10^{-6}	1.6×10^{-5}	5.0×10^{-6}	8.5×10^{-10}	5.1×10^{-7}	1.0×10^{-6}	2.1×10^{-7}	8.1×10^{-6}	3.8×10^{-7}	1.8×10^{-11}
Anaerobic digestion	9.5×10^{-6}	2.6×10^{-10}	1.3×10^{-6}	3.7×10^{-6}	1.4×10^{-6}	2.3×10^{-10}	2.9×10^{-7}	1.2×10^{-7}	2.6×10^{-8}	1.1×10^{-5}	1.8×10^{-6}	8.3×10^{-11}
Energy production	8.5×10^{-5}	2.3×10^{-9}	3.1×10^{-6}	7.4×10^{-4}	2.1×10^{-6}	3.4×10^{-10}	1.7×10^{-7}	1.5×10^{-7}	3.1×10^{-8}	4.6×10^{-6}	1.9×10^{-7}	1.8×10^{-11}
Avoided energy	-3.1×10^{-4}	-8.4×10^{-9}	-1.9×10^{-5}	-3.2×10^{-5}	-7.7×10^{-5}	-1.2×10^{-8}	-2.0×10^{-7}	-2.0×10^{-6}	-4.2×10^{-7}	-3.4×10^{-6}	-2.4×10^{-6}	-1.1×10^{-10}

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Journal of Cleaner Production 380 (2022) 134887

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Environmental benefits of circular food systems: The case of upcycled protein recovered using genome edited potato

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ARTICLE INFO

Keywords:
Life cycle assessment (LCA)
Resource recovery
Circular bioeconomy
Food waste valorization
CRISPR-Cas9

ABSTRACT

Although essential in the human diet, large quantities of available protein are currently lost or under-utilized within the food system, including protein rich side streams from conventional potato starch production. By using the genome editing technique CRISPR-Cas9, conventional starch potato cultivars can be upgraded to facilitate high-value recovery of potato protein fit for human consumption. In turn, this could support the nessecary transition towards more circular food systems. The aim of this study was to assess what environmental benefits could be gained by shifting from conventional protein recovery practice to a novel approach using genome edited potato. Our results, using consequential life cycle assessment, showed that the novel protein recovery scenario provided substantial environmental savings for every ton potato starch produced, with a reduction in global warming impact, terrestrial acidification, land use and ecosystem damage of −720 $kgCO_2eq$, −13 $kgSO_2eq$, −760 m^2a crop eq, and -1.1×10^{-5} species.yr respectively. The potential environmental benefits of using genome edited potato were maintained even when simulating reduced tuber yield, increased production inputs, and substitution of various protein sources. Although currently limited by EU legislation and technical maturity, high-value protein recovery from food side streams holds a promising potential to support sustainable production and circularity within the food system.

1. Introduction

Due to limited planetary resources and global population growth, a transition towards circular food systems will be required to meet the future nutritional needs. Our ability to increase food production is limited by an increasing competition for land and natural resources, often resulting in overexploitation, damage to vital ecosystems, and substantial environmental impact (Rockström et al., 2020; Tian et al., 2021). Resource inefficiencies, under-utilized food side streams and unsustainable dietary patterns are major risk factors for global food security, climate change mitigation and maintained biodiversity (Crenna et al., 2019). To ensure sustainability within future food systems, multi-action approaches are urgently required, ranging from circular supply chains, increased production limits and use of genetically engineered crops, to a shift towards plant-based diets (Godfray et al., 2010; Qaim, 2020). Achievement of these goals require in-depth knowledge and assessment of the supply chain (Vidergar et al., 2021).

Protein from animal and plant sources supplies essential macronutrients to the human diet. With a steadily growing global population, the increased demand for protein must be accommodated without exceeding the planetary boundaries or further jeopardizing the sustainable development goals set by the United Nations (Scherer et al., 2020). A transition towards plant-based diets generally infers lower environmental impact, while also bringing additional health benefits compared with animal-based diets (Willett et al., 2019; Röös et al., 2020; Rysselberge and Röös, 2021). Potatoes are considered among the most important food crops globally (FAO. World, 2021), containing both starch, protein, fiber and trace nutrients. Potato starch, derived via industrial processing where the protein, fibre and nutrient fractions are removed, is often used as an additive to improve stability and texture in for instance sauces, bread and soups, as well as gluten-free and plant-based products. Alongside food applications, potato starch is also used within paper and textile industry. The global market for potato starch was around 3.4 million metric tonnes (tons) in 2020, and is expected to continue to increase (Kowalczewski et al., 2022). In Sweden alone, over 878 000 tons of potato were harvested in 2020, of which around 40% was used to produce potato starch (Wahlstedt, 2020). The remaining protein, fiber and trace nutrients arising as side streams are

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https://doi.org/10.1016/j.jclepro.2022.134887
Received 18 March 2022; Received in revised form 18 August 2022; Accepted 24 October 2022
Available online 28 October 2022
0959-6526/

most commonly reused to produce dietary fiber (Sampaio et al., 2020; Singh et al., 2022; Dey et al., 2021), treated in biorefineries, anaerobic digestion, or evaporated and sold as animal feed or fertilizer (Souza Filho et al., 2017; Caldeira et al., 2020; Grommers et al., 2009). However, increased circularity and re-usability of by-products within the food system is identified as an important step towards sustainable food systems.

Potato protein is one of few the plant-based proteins that supplies a complete amino acid profile (Pęksa and Miedzianka, 2021), with a high biological value that implies high quality and absorption when consumed by living organisms (Camire et al., 2009). Isolated potato protein, in particular the main protein structure *patatin*, can be used in food applications to substitute egg or dairy as it exhibits excellent emulsifying, gelatinizing, and foaming properties (Johansson and Samuelson, 2018; Fu et al., 2020a). Unlike soy, dairy, egg, and wheat, potato protein is free from allergens (Hussain et al., 2021). The current protein recovery practice within starch production involves heat coagulation of an acidified potato liquid, which cause irreversible structural changes to the protein molecules. This approach results in a product disadvantageous for food applications (Stark et al., 2020), as neither the amino acid profile nor the functional properties of the protein are maintained. Alternative recovery methods for maintained protein functionality and nutritional value have been extensively researched (Fu et al., 2020b), but due to high development costs and technical limitations no realistic solution for industrial implementation has yet been found. Another limiting factor is high levels of *glycoalkaloids* naturally present in potatoes, including *α-solanine* and *α-chaconine*, which are chemical compounds acting as a defense system against pests and pathogens. To avoid toxic effects when consumed, an upper threshold of 200 mg *total glycoalkaloids* (TGA)/kg fresh potato is set for Swedish food products (Swedish Food Agency, 2021a). Once removed from the starch flow, TGA accumulate in the by-products and thereby reduces its reusability within food application (Schrenk et al., 2020).

The environmental aspect of implementing recovery solutions that allow available food resources to be utilized according to the highest value possible has been thoroughly stressed in previous studies (Despoudi et al., 2021; Scherhaufer et al., 2020; Ciccullo et al., 2021). With respect to sustainable food systems, the environmental cost of under-utilized side streams could be avoided as these resources are ultimately produced in vain. For potato protein side streams this could be achieved using novel plant breeding methods, such as the mutagenesis *genome editing* (GE) technique *CRISPR-Cas9*. This technique could offer substantial benefits compared with conventional potato breeding, especially considering production yields, macronutrient composition, and specific trait development (Tiwari et al., 2022; Hofvander et al., 2022; Hüdig et al., 2022). Multiple studies show that genome edited food crops can also support fulfilment of the UN's sustainability goals (Rashid et al., 2021; Menz et al., 2020). Traditional *genetic modification* (GM), using *transgenesis*, have been cultivated commercially since the 1990s in some parts of the world, with resistance to pests and diseases, tolerance to herbicides, and increased yields being common traits of GM crops (Brookes and Barfoot, 2020). However, commercial crop development within the EU has so far been hindered by legislation (European Commission, 2021a) and only a few GM crops, including soybeans and rapeseed, are currently approved for use in food production (Swedish Food Agency, 2021b). Polarizing concerns among consumers and policymakers regarding the safety and impact of genome-editing have also limited commercializing, despite that no increased risk compared with conventional plant breeding has been scientifically established (Bauer-Panskus et al., 2020; Turnbull et al., 2021; Pixley et al., 2022). In recent years, the public awareness and acceptance regarding genome edited food crops have advanced notably since the first GE crop entered the open market in 2021 (Waltz, 2021). Many researchers also highlight the long-term consequences of restricted development, and argue that innovations are required to achieve sustainable food production (Herrero et al., 2021; Camerlengo et al., 2022). It is further suggested that the potato crop in particular would benefit greatly from novel plant breeding methods (Halterman et al., 2016), where the CRISPR-Cas9 is considered one of the most versatile tools for crop improvement (Hofvander et al., 2022). Applied within conventional potato starch production, this approach can facilitate improved protein recovery in two main ways: i) by reducing the natural TGA level, and ii) by stabilizing the *patatin* structure and making it less sensitive to heat (Johansson and Samuelson, 2018). In turn, this would enable protein recovery with maintained amino acid functionality and low TGA levels. As considerable quantities of potato protein are produced annually, this could be utilized to support future protein needs.

Assessing the environmental performance compared to current practice is crucial to enable evaluation of the sustainability potential of emerging technologies. The general consensus suggests that novel plant breeding techniques infer lower climate impact and reduced acidification, while also supporting maintained ecosystem functionality (Dastan et al., 2020; Eriksson et al., 2018). At present, the impact categories global warming and acidification are most frequently included in agricultural life cycle assessments (Alhashim et al., 2021), while ecosystem aspects are rarely adequately assessed (Asselin et al., 2020). Multiple studies have emphasized the importance of including land use and ecosystem aspects when assessing food impact (Bartek et al., 2021; Crenna et al., 2020). Despite its scientific and industrial relevance, research is currently lacking with respect to the ecosystem impact of using genome edited crops. Moreover, no previous study has to our knowledge assessed the environmental impact of using CRISPR-Cas9 to facilitate large-scale recovery of high-value potato protein. The aim of this study was therefore to assess and evaluate the environmental performance and potential ecosystem damage of introducing a genome edited cultivar in conventional potato starch production. Our results could thereby provide important support for policy recommendations, further research and development, industry implementation, and increased consumer awareness.

2. Material and method

2.1. Goal and scope

Life cycle assessment (LCA) is a systematic method for quantifying environmental impact during a product's life cycle. Following the ISO-standards, a prospective consequential approach (CLCA) was used to assess the impact shifting to a high-value protein recovery within current potato starch production (Ekvall et al., 2016). A functional unit representing *1 ton potato starch* was selected to reflect the function of the main product, where data from previous research was used to model an industrial-scale recovery process of potato protein. The LCA-software SimaPro 9.3 was used to model the system, using the ReCiPe 2016 (H) method to assess impact at midpoint and endpoint level. Marginal datasets from Ecoinvent 3.8 was used for the background system, while substitution via system expansion was favored over economic or mass allocation in the foreground system (ISO. ISO, 2006). This study assumed a negligible influence of market forces and indirect land transformation to simulate maintained production costs and efficiency during commercial implementation. Potential deviations from current practice are instead addressed in the sensitivity analyses to enable a transparent scenario analyses.

2.2. Description of scenarios

The systems were modeled as two parallel starch scenarios: a conventional *Feed scenario* where the potato protein replaced Brazilian soybean meal, and a conceptual *Food scenario* where potato protein replaced Swedish eggs. Included in the system boundary were production and end-of-life for required inputs, alongside transport and buildings used for processing. Construction and maintenance of additional infrastructure were outside the scope of this study. Sweden was the site location for both scenarios, primarily using site-specific input data. Avoided products due to by-product substitution was assumed to replace equivalent products, based on either nutritional profile or commercial use. Supporting information, including datasets used to model inputs, is available in *Supplementary Material*.

2.2.1. Feed scenario: conventional potato starch process

Potato starch produced in northern Europe often use the Solanum tuberosum L. cv. *Kuras*, with an average macronutrient composition of 75% water, 19% starch, 2% protein, 1.6% fiber, and 2.4% trace nutrients (Godard et al., 2012). The main output is native potato starch, while fiber from potato pulp, protein from potato fruit juice, and agricultural fertilizer refined from potato water are the main by-products (Fig. 1).

An average tuber yield of 50 ton per hectare (ha) was used in this study (Stark et al., 2020), requiring an irrigation input of 30 mm three times per season (L ä nsstyrelsen i V ä stra G ö talands l ä n, 2018) and electricity input of 1062 kWh per ha to power the irrigation pump (Lundgren, 2000). Additional input of insecticides, herbicides, and fungicides was used according to previous research (Ahlmén and Ingvarsson, 2002), alongside input of mineral fertilizers (Kalium, 2021). A total of 18 diesel tractor hours per ha was assumed during cultivation and harvest (L ä nsstyrelsen i V ä stra G ö talands l ä n, 2018). Assuming an average 2% (w/w) deducted tuber loss, the harvested potatoes were transported 30 km from farm to starch factory (Axelsson, 2013). Input data used to model the cultivation, harvest and delivery of potatoes, are listed in Table 1.

During the conveying and cleaning process that initiates starch production, about 1% (w/w) of dirt and soil is removed using fresh water and mechanical scrubbing. The amount of water required was based on previous data (Pingmuanglek et al., 2017), of which 15% was re-circulated from fiber processing (Axelsson, 2013). Following cleaning, the tubers are shredded to release starch granules from the cellular fluid, whereupon the potato pulp is separated from the potato fruit juice (PFJ). Around 5–12 m^3 of PFJ, with a protein content of 30–41% (w/w), can be obtained per ton potato (Karboune and Waglay, 2015). A centralizer and suction process was used to separate starch from PFJ (Lyckeby, 2020), and the remaining starch milk is then refined and dewatered using centrifuges before dried to a final water content of 20% (w/w) (Kot et al., 2020). Energy and heat inputs were used according to previous studies (Lundholm, 2020). All inputs used during native starch production are listed in Table 2.

Table 1
Process data for the *cultivation* process, expressed per 1 ton potato starch.

		Amount	Unit
Planting and cultivation			
Input	Potato seeding	1.80×10^{2}	kg
	N fertilizer	1.57×10^{1}	kg
	K_2O fertilizer	9.59×10^{0}	kg
	P_2O_3 fertilizer	3.57×10^{0}	kg
	Herbicide	0.02×10^{0}	kg
	Insecticide	0.26×10^{0}	kg
	Fungicide	0.04×10^{0}	kg
	Electricity	9.26×10^{1}	kWh
	Water	7.85×10^{1}	m^3
	Diesel	1.01×10^{1}	kg
	Machinery	3.00×10^{-3}	unit
	Transport (to farm)	6.10×10^{1}	km
Harvest and transport			
Input	Machinery	3.00×10^{-3}	unit
	Diesel	6.73×10^{0}	kg
	Transport (from farm)	3.00×10^{1}	km
Output	Harvested tubers	4.36×10^{0}	ton

After separation from the starch flow, the potato pulp is further treated to obtain a fiber product for reuse within food and feed application (Stärkelseproducenter, 2019). Once removed, the PFJ is pumped to a protein recovery facility to remove remaining insoluble components before heat treatment. Using a centrifuge and electrical dryer, the coagulated protein fraction is dried to 80% (w/w) protein content (Johansson and Samuelson, 2018) for reuse in animal feed. An evaporation process is used to refine the potato water before re-used as agricultural fertilizer (Lyckeby, 2020). Energy inputs in by-product processing was used according to previous studies (Lundholm, 2020), and the input data required for each process stage are listed in Table 3.

Potato fiber has similar commercial use as wheat bran in food applications and barley grain in feed production, and was assumed to replace equal amounts these products (Lyckeby, 2020). Potato protein is considered a high-quality alternative to non-GM Brazilian soybean meal

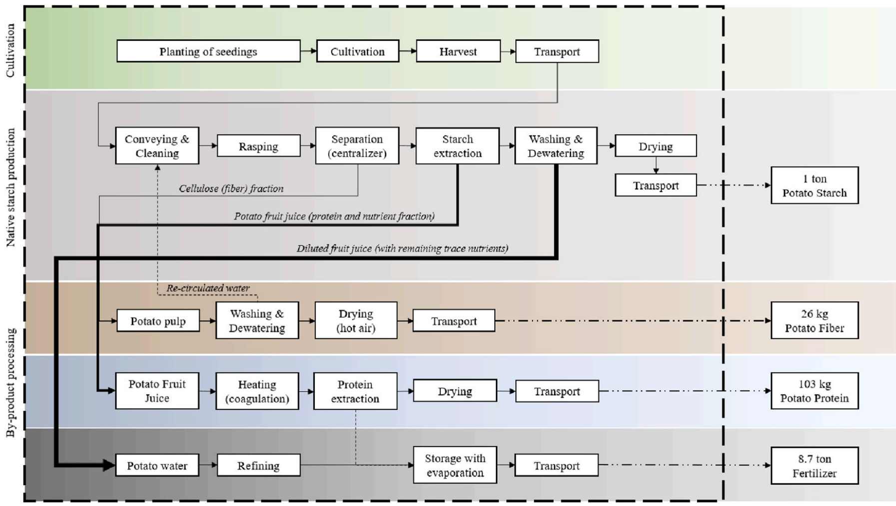

Fig. 1. Illustration of processing required in conventional starch production to produce native potato starch, with potato fiber, protein, and fertilizer as by-products. The dashed line represents the system boundary, and dotted lines show re-circulated flows.

Table 2
Process data for *native starch production,* expressed per 1 ton starch.

		Amount	Unit
Conveying and cleaning			
Input	Harvested potatoes	4.36×10^0	ton
	Water	8.30×10^0	m^3
	Water (re-circulated)	1.34×10^0	m^3
	Electricity	3.14×10^1	kWh
	Heat	3.52×10^1	kWh
	Processing facility	0.10×10^{-6}	unit
Output	Cleaned potatoes	4.31×10^0	ton
	Wastewater	8.35×10^0	m^3
	Solid residues	5.00×10^1	kg
Rasping and separation			
Input	Cleaned potatoes	4.31×10^0	ton
	Water	3.99×10^1	m^3
	Electricity	3.14×10^1	kWh
	Heat	3.52×10^1	kWh
	Machinery	0.19×10^{-3}	unit
	Processing facility	0.10×10^{-6}	unit
Output	Bulk starch milk	4.29×10^1	ton
	Potato pulp	1.36×10^0	ton
Starch extraction			
Input	Bulk starch milk	4.29×10^1	ton
	Electricity	8.72×10^1	kWh
	Heat	3.48×10^1	kWh
	Machinery	0.19×10^{-5}	unit
	Processing facility	0.10×10^{-6}	unit
Output	Starch milk	2.47×10^0	ton
	Potato fruit juice	3.66×10^1	ton
	Wastewater	0.14×10^0	m^3
Starch purification			
Input	Starch milk	2.47×10^0	ton
	Water	7.77×10^0	m^3
	Electricity	1.10×10^{-1}	MWh
	Heat	1.50×10^{-1}	MWh
	Machinery	0.19×10^{-5}	unit
	Processing facility	0.10×10^{-6}	unit
Output	Native potato starch	1.00×10^0	ton
	Potato water	9.24×10^0	m^3

Table 3
Process data for *by-product processing,* expressed per 1 ton potato starch.

		Amount	Unit
Fiber processing			
Input	Potato pulp	1.40×10^0	ton
	Electricity	1.93×10^1	kWh
	Heat	7.33×10^1	kWh
	Pulp processing factory	0.10×10^{-6}	unit
	Transport	6.10×10^0	km
Output	Potato fiber	2.61×10^1	kg
	Water (re-circulated)	1.30×10^0	m^3
Protein production			
Input	Potato fruit juice	3.66×10^1	ton
	Electricity	1.56×10^1	kWh
	Heat (steam)	5.30×10^1	kWh
	Heat (gasol)	5.30×10^1	kWh
	Protein production facility	0.20×10^{-6}	unit
	Transport	6.00×10^1	km
Output	Potato protein	1.03×10^2	kg
	Wastewater	1.82×10^1	m^3
	Potato water	1.83×10^1	m^3
Fertilizer refining			
Input	Potato water	2.76×10^1	m^3
	Electricity	3.45×10^1	kWh
	Heat	2.70×10^1	kWh
	Evaporation facility	0.10×10^{-6}	unit
	Transport	3.00×10^1	km
Output	Potato fertilizer	8.72×10^0	ton
	Wastewater	1.88×10^1	m^3

in Swedish feed production (Hermansson, 2013), and the protein fraction of 50% (w/w) (Ibáñez et al., 2020; Spiller et al., 2020) was assumed 1:1 replaceable with potato protein. The potato water was assumed to replace mineral nitrogen and phosphorus fertilizers (Spiller et al., 2020). Inputs used to model the substituted products via system expansion are listed in Table 4.

2.2.2. *Food scenario: conceptual process using genome edited potato*

The main difference from current practice is that the *Food scenario* allow reuse of potato protein within food applications, and therefore the only modeling difference is substituted protein. Since potato protein is of high nutritional value comparable to animal-based proteins and has similar functional properties to eggs (Hussain et al., 2021), this study assumed 1:1 replacement of protein from eggs with an average protein content of 12.5% (w/w). A separate model was created to describe Swedish egg production for the substituted protein, see *Supplementary Material*, and inputs used to model the substituted protein via system expansion are listed in Table 5.

2.3. *Sensitivity analysis*

The first sensitivity analysis sought to evaluate uncertainties related to input data, where different fertilizers potentially influence environmental impact (Hanserud et al., 2018). This was addressed by shifting to organic fertilizers during cultivation. The two subsequent analyses addressed uncertainties related prospective genome editing techniques. One of the two main ways in which CRISPR-Cas9 can facilitate improved protein recovery is by stabilizing the protein structures, which would infer a recovery process requiring less energy and water inputs. This was accounted for by simulating 30% decreased demand for electricity, heat, and water during starch and protein production. The other way in which CRISPR-Cas9 can be used is by reducing the amount of TGA present (Johansson and Samuelson, 2018), and a potential consequence of reduced TGA levels could be higher use of pesticides to offset a lower tuber defense. This uncertainty was addressed by doubling the pesticide input required. Another relevant aspect for the LCA method is sensibility to production yields, which was evaluated by simulating a decreased tuber yield which ultimately inferring a 20% increase of inputs during all production stages. Another limitation is substitution sensitivity, as this methodological approach might influence the environmental performance (Vadenbo et al., 2017). This was addressed by simulating replacement of alternative protein sources in feed and food.

3. Results

3.1. *Environmental impact and ecosystem damage*

With respect to environmental impact, the result show that the *Food scenario* inferred lower impact for 13 of 18 midpoint indicators compared with the conventional system. Using potato protein to

Table 4
Process data for the *substituted products,* expressed per 1 ton potato starch.

		Amount	Unit
Substituted fiber			
Input	Potato fiber	2.61×10^1	kg
Avoided product	Wheat bran	-1.31×10^1	kg
	Barley grain	-1.30×10^1	kg
	Transport (to industry)	-6.00×10^1	km
Substituted protein			
Input	Potato protein	1.03×10^2	kg
Avoided product	Soybean meal (Brazil)	-1.74×10^2	kg
	Transport (Brazil)	-1.19×10^3	km
	Transport (Norway)	-1.07×10^4	km
	Transport (Sweden)	-5.92×10^2	km
Substituted fertilizer			
Input	Potato fertilizer	8.70×10^0	ton
Avoided product	N fertilizer	-2.60×10^0	kg
	P fertilizer	-0.50×10^0	kg
	Transport (to farm)	-6.00×10^1	km

Table 5
Process data for *substituted protein,* expressed per 1 ton potato starch.

		Amount	Unit
Substituted protein			
Input	Potato protein	1.03×10^{2}	kg
Avoided product	Egg (Sweden)	-6.62×10^{2}	kg
	Transport	-6.00×10^{1}	km

substitute eggs over soybean meal changed the global warming result from a negative impact to global warming savings, while also close to halving the land use impact and reducing terrestrial acidification by over five-fold (Table 6). The *Food scenario* was also found to infer lower marine eutrophication and fine particulate matter formation, while the Feed scenario resulted in lower mineral resource scarcity and toxicity. A negative value indicates reduced environmental impact and origin from substitution.

The *Food scenario* also inferred lower damage to 8 of 12 endpoint categories compared with the conventional *Feed scenario*. For every ton starch produced, the *Feed scenario* resulted in -6.8×10^{-6} species.yr and the *Food scenario* -1.8×10^{-5} species.yr. This translates to over two-fold more favorable conditions with respect to biodiversity, since a negative value for ecosystem damage indicate mitigated loss of species. With respect to ecosystem damage, the results showed that the main contributing factors for both scenarios were global warming and land use, while in the *Food scenario* there was also a considerable contribution from reduced terrestrial acidification (Fig. 2).

3.2. Sensitivity analysis

Shifting to the *Food* scenario resulted in higher environmental savings with respect to global warming, land use, and terrestrial acidification even when simulating increased pesticide use, organic fertilizers, reduced processing inputs, and decreased tuber yield. Only a slight change was observed when assessing the sensitivity of data parameters, indicating that these were not a major source of uncertainty in this study. On the other hand, the protein product substituted was shown to give higher uncertainty. Overall, the results indicated that the highest biodiversity and environmental savings, with respect to climate impact, acidification, and land use, were obtained when the recovered potato protein replaced animal-based protein sources, such as eggs and dairy (Fig. 3).

Table 6
Environmental impact per 1 ton potato starch produced.

		Feed scenario	Food scenario	Difference (absolute value)
Global warming	kg CO_2 eq	1.8×10^{2}	-5.4×10^{2}	-7.2×10^{2}
Stratospheric ozone depletion	kg CFC_{11} eq	-1.1×10^{-3}	-4.1×10^{-3}	-3.0×10^{-3}
Ionizing radiation	kBq Co-60eq	1.1×10^{1}	2.8×10^{0}	-8.3×10^{0}
Ozone formation. Human health	kg NOx eq	1.0×10^{0}	4.8×10^{-1}	-5.3×10^{-1}
Fine particulate matter formation	kg $PM_{2.5}$ eq	2.1×10^{-1}	-9.0×10^{-1}	-1.1×10^{0}
Ozone formation. Terrestrial ecosystems	kg NOx eq	9.7×10^{-1}	5.2×10^{-1}	-4.5×10^{-1}
Terrestrial acidification	kg SO_2 eq	1.9×10^{0}	-1.1×10^{1}	-1.3×10^{1}
Freshwater eutrophication	kg P eq	-3.5×10^{-1}	2.1×10^{-1}	5.6×10^{-1}
Marine eutrophication	kg N eq	-2.4×10^{-1}	-1.1×10^{0}	-8.5×10^{-1}
Terrestrial ecotoxicity	kg 1.4-DCB	1.0×10^{4}	1.2×10^{4}	1.4×10^{3}
Freshwater ecotoxicity	kg 1.4-DCB	5.5×10^{1}	5.6×10^{1}	3.7×10^{-1}
Marine ecotoxicity	kg 1.4-DCB	7.5×10^{1}	8.2×10^{1}	7.6×10^{0}
Human carcinogenic toxicity	kg 1.4-DCB	4.4×10^{1}	3.8×10^{1}	-6.4×10^{0}
Human non-carcinogenic toxicity	kg 1.4-DCB	1.2×10^{3}	9.4×10^{2}	-2.7×10^{2}
Land use	m^2a crop eq	-8.7×10^{2}	-1.6×10^{3}	-7.6×10^{2}
Mineral resource scarcity	kg Cu eq	2.9×10^{0}	4.2×10^{0}	1.2×10^{0}
Fossil resource scarcity	kg oil eq	2.0×10^{2}	1.6×10^{2}	-4.1×10^{1}
Water consumption	m^3	1.1×10^{2}	9.5×10^{1}	-1.6×10^{1}

4. Discussion

One of the key findings in the present study was that using genome edited crops to facilitate high-value protein recovery from potato starch side streams could infer considerable environmental savings compared with the conventional recovery practice. The conceptual *Food scenario* was shown to reduce the environmental impact for every ton starch produced, especially considering global warming, terrestrial acidification, and land use. These midpoint indicators also had the highest contribution to ecosystem damage, likely since the ReCiPe method applies a constant characterization factor from midpoint to endpoint level (Huijbregts et al., 2017). The environmental cost of cultivating and production showed a negligible contribution to the overall impact compared with the environmental savings enabled by replacing protein products. Compared with current practice, the *Food scenario* also resulted in over two-fold the ecosystem savings, illustrating considerable benefits of genome edited potato protein recovery with respect to biodiversity. These results can be considered valid provided that no adverse trait or quality consequences emerge from editing the potato genome. When assessing impact of emerging technologies, it is especially important to consider uncertainties related to technological maturity and limited representation in available datasets. The benefits of recovering available resources and re-circulating them back to the food system was evaluated in sensitivity analyses, where substituting food over feed unequivocally inferred higher ecosystem and environmental savings. This is in line with previous findings (Scherhaufer et al., 2020; Moreno-González and Ottens, 2021; Scuderi et al., 2021), which suggest that circular recovery of high-value protein could reduce the environmental impact related to food and thereby support production within planetary boundaries.

4.1. Environmental impact of protein recovery in potato starch production

At midpoint level, the *Food scenario* resulted in lower impact for the majority of all impact categories. Cultivation, native starch processing and substituted protein were the main environmental hotspots with respect to global warming, acidification, land use, and ecosystem damage for both scenarios (Fig. 3). The cultivation process relies on large material inputs such as fertilizers, pesticides, water and fossil fuels to ensure high returns. Production of these inputs and combustion-related emissions are probably the main cause for global warming impact and land use, while terrestrial acidification could also origin from ammonia volatilization after fertilizer application (Alhashim et al., 2021). Addressing the data uncertainty related to specific products, the sensitivity analyses of using organic fertilizers showed comparable impact to mineral fertilizers in this study. A similar conclusion can be drawn for reduced need for energy and water inputs, as the result showed that neither fertilizer nor input quantities were a major source of uncertainty. On the contrary, the results instead confirmed that tuber yield was the most sensitive data parameter in this study. This result was somewhat expected since the LCA method originally was developed for industrial processes aiming to reduce impact per production unit, thus the results implicitly tend to favor systems with high production yields. Important to note is that these results should primarily be considered valid for

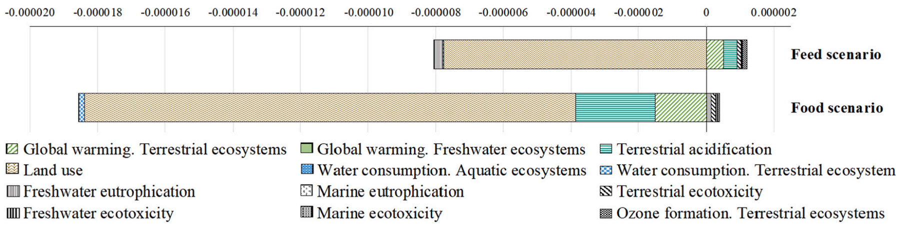

Fig. 2. *Illustration of the contribution for each endpoint indicator to the total ecosystem damage (species.yr) assessed for the* Feed *and* Food *scenarios. A negative value indicates contribution to mitigated damage to ecosystems, primarily due to substitution.*

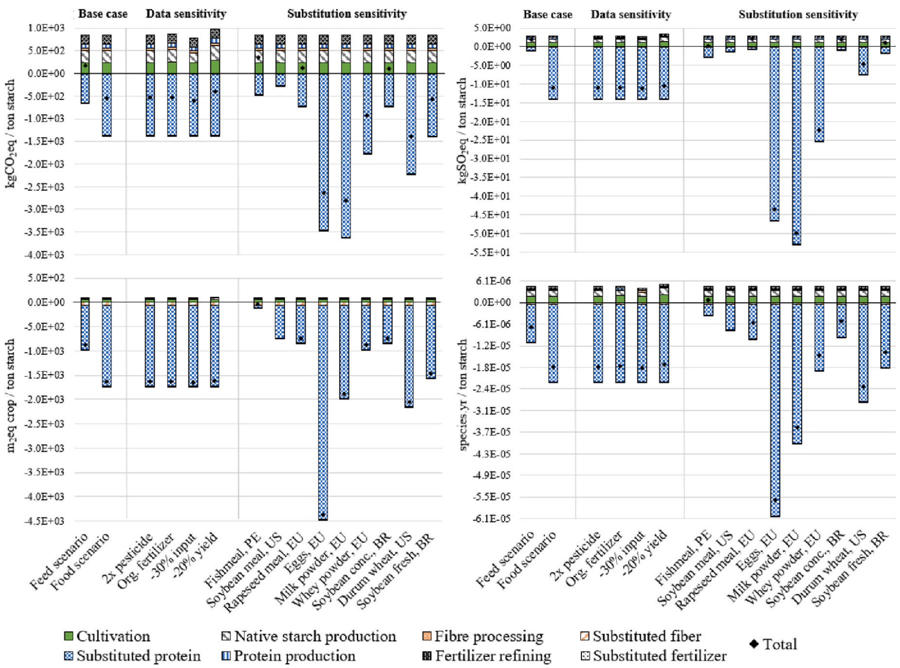

Fig. 3. Sensitivity analysis results with respect to global warming potential, terrestrial acidification, and land use at midpoint level, and ecosystem damage at endpoint level. The contribution of each process to total impact is indicated.

scenarios implemented within current potato starch industry, as major deviations in infrastructure or management was outside the scope of this study. Our results indicate that production yields, climate impact, and land use tend to govern environmental impacts obtained using the LCA method, which is also consistent with previous research (van der Werf et al., 2020). During starch production and by-product processing, the main factors contributing to global warming and terrestrial acidification were electricity inputs and combustion-related emissions from petroleum heating. Since this study assumed input of Swedish electricity mix produced from less than 45% fossil fuels, a low contribution to these impact categories was expected. Compared with the relatively low impact from electricity and water, the impact of buildings was larger than expected, especially since this aspect is often omitted in LCA studies. With respect to land use, the largest contribution originated from inclusion of processing facilities. Although, as the same buildings were used in both scenarios this process related uncertainty is avoided when comparing the systems.

The main sensitivity for the *Feed* and *Food scenario* was substitution of protein products. Higher savings of substituting animal-based was expected (Röös et al., 2020), but particular care was taken to validate the impact related to substitution of eggs and soybean meal as substitution was the main source of sensitivity. Previous studies (Rysselberge and Röös, 2021; Moberg et al., 2019) suggest that Swedish eggs infer a climate impact of 2 $kgCO_2eq$ per kg, which corresponds to about −1300 $kgCO_2eq$ per ton starch when using a 12.5% protein content. This is in line with the results in this study (Fig. 3). The environmental impacts of non-GM and GM soybean meal imported to Sweden has been shown to infer climate impact of 845 $kgCO_2eq$ per ton and 609 $kgCO_2eq$ per ton respectively (Eriksson et al., 2018). With 50% protein content, this translates to climate savings of −282 $kgCO_2eq$ for non-GM varieties and −203 $kgCO_2eq$ for GM per ton starch. These results are in the same order of magnitude as the climate impact of Brazilian non-GM soybean meal and GM soybean meal from US (Fig. 3), although the numerical value for non-GM soybean meal is almost double to that previously found. Plausible reasons for this difference are newer datasets used to describe production and transport, and different assumptions regarding soybean yields. Moreover, the assumed protein content for soybean meal in this study slightly higher than suggested in other studies (Eriksson et al., 2018; Ibáñez et al., 2020), which likely influenced the results as lower protein content per mass unit requires larger product volumes. Moreover, to our knowledge no previous LCA study has accounted for the biological value (BV) of protein products. In the present study, when accounting for the BV, the climate savings of substituted soybean meal (BV 84) would be just over −700 kgCO_2eq per ton starch and the climate savings for substituted Swedish eggs (BV 100) would be around −1400 kgCO_2eq per ton starch (see *Supplementary Material*). Ultimately, accounting for biological value or protein quality could infer higher environmental and ecosystem savings than previously assessed, which is in line with previous research findings (Sonesson et al., 2017). Overall, the result for climate impact of eggs and soybean meal can be considered supportive of previous findings for Swedish conditions.

Another important finding was the importance of including acidification and land use impact when evaluating the environmental performance and ecosystem damage of a food product. The results revealed some potential trade-offs, especially considering climate impact and ecosystem damage. The assumption that potato protein fully replace non-GM Brazilian soybean meal could be considered as an optimal scenario for feed substitution, since the other protein sources in feed inferred considerably lower environmental and ecosystem savings per ton starch. On the contrary, replacing Swedish eggs was found less favorable compared with e.g. replacing European eggs, and thus does not pose as an optimal scenario from an environmental point of view. Moreover, substituting milk powder was the most favorable alternative with respect to climate impact and acidification, while substituting EU eggs was the most favorable with respect to land use and ecosystem damage. Trade-offs like these might cause competition and conflicts between which UN sustainable development goals should be prioritized, demonstrating the importance of addressing potential trade-offs when evaluating different production processes. Similarly, our results indicated that the benefit of avoided mineral fertilizer production did not exceed the environmental cost of refining potato water via evaporation, highlighting a potential need to evaluate the best recovery practices for potato water from the current starch industry.

4.2. Limitations and future outlook

The present results imply that climate impact, acidification, and land use could be suitable indicators for ecosystem damage. However, no LCIA method can fully account for ecosystem damage caused by e.g. direct pesticide application (Huijbregts et al., 2017). This method uncertainty was evident as global warming impact inferred higher damage to ecosystems than additional use of toxic chemicals (Fig. 3), and emphasize the importance of further method development. Another limitation of prospective CLCA studies is the aspect of market effects and land transformation. Even though potato protein is considered a high-value alternative to soybean meal, total replacement of soybean meal is not yet realistic due to competitive prices, limited availability, and current legislation (Hermansson, 2013). Non-GM soybean meal is generally more expensive than GM alternatives (Eriksson et al., 2018), which ultimately infer higher production costs for farmers operating in countries with zero GM tolerance. Thus, a potential outcome of commercialized GE potato could be increased competition with GM crops rather than non-GM varieties. Moreover, if the potato protein were to be upcycled to food production, alternative protein sources for feed would be required to fill the gap. This secondary effect was not covered in the present study, but should be considered in future research as it might reveal important sustainability dimension of upcycled food side streams. Another consideration is that the price of Swedish eggs, which is highly influenced by the production practice (e.g., eco-friendly feed or free-range hens), while the nutritional value is fairly constant. The market value of eggs is therefore difficult to compare from a strict price perspective, since social and environmental aspects also affect demand for this product.

Increased macronutrient recovery, circular food systems, and reduced environmental and ecosystem impacts are fundamental global objectives. Innovations to enable industrial implementation and commercialization would initially be dependent on economic support via research investments, together with legislation to facilitate development. From a sustainability point of view, the social acceptance and economic values related to GE food crops also need to be considered (Peschel and Aschemann-Witzel, 2020). On the international market, genome edited crops have avoided much of the negative controversy related to GM organisms, ultimately since the two techniques are fundamentally different. Although a negative consumer attitude has previously hindered further commercialization, research has observed an increasingly positive consumer attitude towards novel GE food crops (Ramadas et al., 2021). In 2021, European Commission concluded that current legislation for GE food crops is not fit for its purpose, and a modernized policy is currently under discussion (Pixley et al., 2022; European Commission, 2021b).

At present, potato protein can be recovered without genome edited crops, but it is not fit for human nutrition. Since the market value of protein is strongly affected by its purity and nutritional composition, the long-term payoff with respect to profitability and circularity within food systems could be greater with GE potato. It is reasonable to assume that a certain amount of competition with existing products may initially occur when new products are introduced to the commercial market, and for potato protein this might influence production of eggs and imported soybean protein, but could also replace dairy or gluten in sauces, bread, and plant-based products. However, a commercialized potato protein recovered using genome edited crops should initially be considered as a complementary protein source, rather than a competing protein product. Using this available resource at a higher value could also provide an economic advantage for producers and the industry (Scuderi et al., 2021), especially since recovered plant-based protein is predicted to play an important role in future protein production. Similarly, recovered potato protein would further support resilient agriculture and increased food security, as reduced ecosystem damage and maintained biodiversity are cornerstones of resilience to external stressors such as pests, climate change and extreme weather (Colgrave et al., 2021). Further research, development work, industry implementation, and legislation supporting innovations will play a vital role in enhancing future protein production.

4.3. Key recommendations

Maintaining a stable and sustainable food supply, while balancing population growth and limited natural resources, is one of our most urgent global challenges. Since the feed and food sector is highly dependent on imported plant protein, recovering potato protein from local starch side streams could bring meaningful benefits with respect to national self-sufficiency, local entrepreneurship, and fulfilment of environmental objectives. Arguably, adequate policy recommendations and legislation should accommodate these aspects. Moreover, ensuring Nature's ability to provide the resources needed for global food security requires urgent and substantial actions that comply with Agenda 2030 and UN sustainable development goals. As of yet, researchers have failed to identify a stand-alone solution for how to sustainably feed 9 billion people. On the contrary, the main way forward is a broad range of simultaneously adapted solutions aimed to ensure long-term sustainability within the food system. Maintained production yield and enabling protein upcycling to the food system were identified as two of the main factors that should be prioritized in further development of genome edited potato. To fully utilize the benefits identified, there is an urgent need for sufficient research initiatives and substantial policy interventions. The future success will also depend on legal boundaries set by the EU and consumer acceptance. Positive consumer response to genome edited foods can be promoted by government and policy recommendations, while reduced prices, transparent and visible research results, alongside targeted market communication can positively influence consumer acceptance. Such actions could be justified based on the environmental savings, future demand for plant-based protein, and known health benefits. Arguably, all means available actions should be considered to reduce climate impacts, maintain biodiversity, and increase circularity within food production. If not enough is done in time, our conditions for sustainable production and consumption will be further jeopardized and inevitably impact all life on Earth. The ultimate question is therefore whether the benefits of recovering available resources from food side streams exceed the current limitations. This study showed that the environmental benefits of using genome edited crops to facilitate high-value potato protein recovery clearly exceed the environmental costs.

5. Conclusions

This assessment showed that a genome edited potato cultivar created using CRISPR-Cas9 could facilitate high-value protein recovery with considerable environmental savings and avoided damage to ecosystems. Compared with current practice, the new technology resulted in lower environmental impact for 13 of 18 midpoint impact categories per ton starch produced, including over three-fold reduced global warming impact, more than five times lower terrestrial acidification, and about half the land use impact. Shifting to high-value recovery of potato protein also halved the damage to ecosystems, ultimately supporting maintained biodiversity and sustainability within agriculture production. Although the sensitivity analysis showed a model sensitivity towards parameters related to substituted protein products, re-circulating potato protein to the food system still comprised the most favorable scenario from an environmental perspective despite identified sensitivities. If current barriers can be overcome with the prospect of environmental savings, this available plant-based food resource can facilitate increased protein production per cultivated hectare, and thus drive the transformation towards higher circularity and sustainability within the food system.

CRediT authorship contribution statement

L. Bartek: Conceptualization, Methodology, Software, Validation, Investigation, Data curation, Writing – original draft, Writing – review & editing, Visualization, Project administration. **N. Sundin:** Methodology, Resources, Validation, Writing – original draft. **I. Strid:** Conceptualization, Validation, Supervision, Writing – review & editing. **M. Andersson:** Conceptualization, Resources, Writing – review & editing. **P-A. Hansson:** Writing – review & editing, Supervision, Validation. **M. Eriksson:** Conceptualization, Validation, Writing – review & editing, Supervision, Project administration, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

This study was funded by the Swedish Research Council for Sustainable Development (FORMAS).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2022.134887.

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III

Sustainable Production and Consumption 55 (2025) 51–62

Contents lists available at ScienceDirect

Sustainable Production and Consumption

journal homepage: www.elsevier.com/locate/spc

The power of prevention and valorisation – Environmental impacts of reducing surplus and waste of bakery products at retail

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ARTICLE INFO

Editor: Prof. Shabbir Gheewala

Keywords:
Take-back agreement
Life cycle assessment
Material flow analysis
Sustainable production
Resource recovery

ABSTRACT

The wastage of edible food still remains a major global challenge, despite its well-known consequences. Although bread and bakery products are identified as some of the most frequently wasted foods, the amounts generated and the pathways used to manage this surplus are often unknown. To support sustainable food systems, there is an urgent need to identify how much surplus is generated along the supply chain, including both sweet and savoury products, and to map how this resource is managed. The aim of this study was to quantify the surplus and waste of baked goods in Sweden, alongside mapping the pathways used for managing unsold bread generated at the supplier-retailer interface. Life cycle assessment, considering 16 midpoint indicators and three endpoint indicator, was used to assess the environmental benefits of reducing bakery product surplus. The results reveal that nearly 180 000 tonnes of baked goods are wasted annually in Sweden. Roughly 51% is generated at the supplier-retailer interface, particularly considering bread sold under take-back agreements where 14% of production becomes surplus. Only 2% of this surplus is recirculated to the food system, while the majority is instead used in energy production. Scenario analyses, including nine scenarios designed to capture various innovations to reduce surplus, demonstrated that prevention and valorisation strategies, such as data sharing and price reductions, have the greatest potential for reducing waste and environmental impact. Prevention could result in up to ten times lower climate impact per kg bread. The findings offer valuable insights for future research on sustainable food systems, and can act as practical guidance for industry actors, stakeholders, and policymakers to implement waste-reduction strategies that promote sustainable, resource-efficient food systems.

1. Introduction

An imbalance between production and consumption of food, inevitably leading to surplus, has been identified as a common cause of waste generation at retail level. Despite the known risks attributed to food waste, global wastage of edible food is a major challenge at all stages of the supply chain. Bread is one of the most frequently wasted food products in many parts of the world (Brancoli et al., 2019; Dymchenko et al., 2023; WRAP, 2023), resulting in considerable global issues with environmental, economic, and social consequences (United Nations Environment Programme, 2024). Generation of bread waste is often linked to production of *surplus bread*, i.e., retail bread that remains unsold and is removed from shelves while still perfectly suitable for human consumption. This food resource could in fact be recovered using circular management pathways, such as *prevention* or *valorisation*. Prevention involves measures that reduce generation of food waste at source, while valorisation involves measures to recover or reuse the resource in, for example, new products, animal feed or energy production. These two approaches are usually ranked according to the food waste hierarchy (Papargyropoulou et al., 2014), indicating the priority of action for policy and action against food waste (Giordano et al., 2020).

The benefits of circular food systems have been thoroughly demonstrated in supporting sustainable use of resources and reduce stress on

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https://doi.org/10.1016/j.spc.2025.01.013
Received 29 July 2024; Received in revised form 21 January 2025; Accepted 23 January 2025
Available online 29 January 2025

planetary boundaries (Rockström et al., 2020; van Zanten et al., 2023). The bread supply chain in Sweden exhibits a degree of circularity, particularly through the implementation of *take-back agreements* (TBA). These agreements operate in a reverse supply chain, holding the supplier financially responsible for unsold products, including collection and disposal (Brancoli et al., 2019). Surplus bread is an abundant, inexpensive and under-utilised resource that could be avoided or recovered via valorisation pathways. However, while TBA can be viewed as a step toward circularity by allowing valorisation of unsold products, this model has also been identified as a risk factor in generating high volumes of bread waste (Eriksson et al., 2017). In Sweden alone, previous research suggests that around 80 000 tonnes of bread are wasted along the supply chain (Brancoli et al., 2020). Wasting surplus bread represents a considerable economic loss for producers, retailers and consumers, but also has a devastating environmental impact through e.g. increased global warming, biodiversity loss, and depletion of natural resources (Crenna et al., 2019; Bergström et al., 2020). Moreover, previous waste quantifications on the Swedish bread system have mainly focused on savoury products, while surpluses of sweet products produced and distributed in parallel have not yet been accounted for. Although prevention and valorisation have attracted increasing scientific and commercial attention in recent years, a substantial research gap still remains regarding the efficacy of different pathways compared with current practice, especially considering resources circulated back to the food system.

The aim of this study was to map the volume of surplus bakery goods generated at the supplier-retailer interface in Sweden and to identify the pathways currently used for managing surplus bread. Using *life cycle assessment* (LCA), the environmental aspect of current and future management of surplus bakery goods was then assessed. The potential benefits of multiple innovations, either within or as a consequence of changes to the current business model, were modeled in scenarios using savoury bread sold under TBA in Sweden as the base case. The goal of this work was to evaluate the outcomes of different prevention and valorisation pathways, in order to support companies in the bakery sector and policymakers in choosing the best-performing options for management of surplus bread.

2. Literature review

The annual wastage of food at retail in high-income countries is estimated to amount to 13 kg per capita, with Swedish retail wasting 9 kg per capita. This translates to roughly 89 000 tonnes at retail, of which 15-30% is estimated to consist of surplus bread (Brancoli et al., 2019; Swedish Environmental Protection Agency, 2024). At retail level, high on-shelf availability and providing a broad selection of products are often prioritised with respect to bread, both due to strong competition and to ensure customer satisfaction (Ghosh and Eriksson, 2019; Riesenegger and Hübner, 2022). However, this have been found to increase overstocking, leading to economic pressures and logistical challenges with unsold products (Cicatiello et al., 2020). Many retailers acknowledge that they are faced with a trade-off situation between providing high availability of products and the environmental, social, and financial burdens related to unsold products. Although on-farm losses and household food waste generally comprise greater quantities, retailers and suppliers have a unique influence on both upstream and downstream food waste generation (Mena et al., 2011).

The generation of surplus bread and bakery products at the supplier-retailer interface has been quantified in several previous studies, though often using different methods, countries of origin, comparison bases, or system boundaries (Goryńska-Goldmann, 2022; Soni et al., 2022). In previous studies in Sweden, Brancoli et al. (2019) combined primary data from 380 stores and a major bakery with national statistics and literature findings to quantify bread waste, while Hildersten et al. (2025) used a qualitative interview approach in collaboration with an industry partner to map their production of pre-packaged rye bread sold under TBA. In their study quantifying surplus bread in Italian small-scale bakeries, Pietrangeli et al. (2023) used a combination of direct measures of surplus bread through daily diaries and calculated values for economic losses at the bakery level. Using individual in-depth interviews with industry experts, Goryńska-Goldmann et al. (2021b) collected qualitative data on average losses for the baking and confectionery industry in Poland, while Riesenegger and Hübner (2022) sourced data for their qualitative study on reducing food waste at retail in Germany by conducting face-to-face interviews with managers of seven case companies.

Various incentives to reduce retail food waste have also been researched, with results highlighting great potential if these actions can be directly influenced by companies and suppliers. Sharing data between retailers and suppliers can allow for more efficient ordering and forecasting (de Moraes et al., 2020), which, in turn, can reduce overproduction and surplus at retail (Canali et al., 2017). Insufficient cooperation and coordination among actors in the supply chain have also been identified as relevant risk factors for bread waste (Cicatiello et al., 2020). Dynamic pricing, which allows retailers to reduce prices depending on external factors such as best-before date or imperfect products, can considerably reduce bread waste generated at retail (Sanders, 2024). In an explorative study on reducing food waste in Germany, Riesenegger and Hübner (2022) showed how better planning of retail operations, such as optimal management of shelving and reduced assortment size, could reduce surpluses. Goryńska-Goldmann et al. (2021a) recommended multiple valorisation pathways for surplus bakery products, including reduced prices, food donations, and animal feed. In a review assessing the effectiveness of the food waste hierarchy in increasing resource use efficiency, Redlingshöfer et al. (2020) concluded that when stakeholders dependent on sales are also responsible for waste management, reuse and recycling methods are often prioritised over prevention. This is likely due to the cost-effectiveness of these methods in relation to potential loss of sales. On the other hand, many benefits of waste reduction have been highlighted (CEC, 2019), including cost savings for stakeholders, reduced environmental impacts, improved food security, and promotion of a circular economy by adding value to resources.

Previous research suggests that TBA can be a risk factor for surplus generation at retail (Muzivi and Sunmola, 2021), and that this practice may reduce the power and incentives of retailers to develop and implement waste-reducing actions (Eriksson et al., 2017). Similar conclusions were reached in later studies by Brancoli et al. (2019) and Ghosh and Eriksson (2019). However, one of the benefits of TBA is the separate collection of surplus bread, which can be directed toward more high-value valorisation than is possible with mixed waste streams. Identifying how and under what conditions future pathways can contribute to reduced bread waste, with or without TBA in place, is therefore of high scientific relevance. According to Economou et al. (2024), monitoring tools, including LCA, are indispensable for mapping surplus hotspots and tracking the impact of waste-reducing actions. Using LCA, Brancoli et al. (2020) showed that prevention of surplus bread yields the highest environmental benefits, followed by reuse as food, either directly or following conversion, while feed and energy production are less favourable. In later work, Brancoli (2021) concluded that the current return systems for surplus bread in Sweden could serve as a foundation for further sustainability development, e.g., by implementing alternative reuse pathways according to the higher priority levels in the food waste hierarchy. However, current valorisation pathways for food waste, including surplus bread, tend to be directed toward energy production rather than human consumption (Johansson, 2021). These solutions align with the lower-priority levels for managing food waste (Papargyropoulou et al., 2014), such as anaerobic digestion, conversion into biofuels, and incineration. One important benefit with these solutions is their high technological readiness, which facilitates quick implementation.

On evaluating the life cycle of surplus food generated in French retail, Albizzati et al. (2019) concluded that the sector should prioritise redistribution through donations and conversion to animal feed over anaerobic digestion and incineration, due to the environmental benefits and economic gains. This was reiterated by Svanes et al. (2019), who further emphasized the environmental benefits of prevention related to management of surplus bread. Assessing the impact of bread logistics, Weber et al. (2023) found that the long transport distance between suppliers and retailers in Sweden is the main climate hotspot, more so than TBAs as a business model. Many studies have demonstrated the potential to utilize bakery surplus in food production, including food donations (Sundin et al., 2023) and upcycling into breadcrumbs (Samray et al., 2019), fungal food products (Brancoli et al., 2021) or beer (Coelho et al., 2024). Jung et al. (2022) used bread as a feedstock in algae cultivation, while Siddique et al. (2024) mapped the life cycle impact of multiple pathways for valorisation into animal feed. Although previous research has demonstrated the benefits of bread valorisation, Corsini et al. (2023) emphasized the lack of research evaluating the success factors influencing prevention actions at the retail level. A similar conclusion was also presented by Kumar et al. (2023), who pointed out that current literature accounting for environmental impacts of valorisation pathways for bread is very limited. Furthermore, research with a lifecycle perspective of the entire bread supply chain, including baking, distribution, retail, and management of surplus products, is urgently required to identify how, and under what conditions, different prevention and valorisation pathways should be used. Siddique et al. (2024) suggested that a broad selection of impact categories should be included in LCA studies, along with the benefit of any avoided emissions due to prevention or valorisation. Despite the acknowledged limitations of ecosystem coverage in LCA, Crenna et al. (2019) showed the importance of also accounting for ecosystem impact in future food system research. Goryńska-Goldmann (2022) further concluded that there is an urgent need for joint actions by suppliers and retailers to mitigate waste generation, preferably through prevention and high-value valorisation.

3. Material and method

3.1. Mapping of Swedish bakery products

To distinguish the different main types of bread and baked products sold in Sweden, the definitions used by the Swedish Board of Agriculture (2022) were applied in this study. Bread, as a product, was considered to be the baked result of a simple dough containing flour (wheat and rye), water, salt and rising agent. Similar ingredients, but in different quantities, are often used to produce sweet products, with the addition of sugar. Two main categories of soft bread are sold at Swedish retail: *pre-packed* soft bread products, which are produced by bakeries and transported to retail where they are sold in plastic bags; and *non-packaged* or *bake-off* products that are delivered as industrially produced doughs or semi-baked products, which are baked after delivery, and sold individually at retail or convenience stores. One of the major differences between these products is the distribution system, where a large proportion of pre-packed products, but not bake-off products, are distributed under TBA. Within the sub-category of pre-packaged goods, retailers also produce their own *private label products* or import from other countries, which are distributed without TBA. Furthermore, the pre-packed category includes sweet and savoury baked goods that have been exposed to air or heat to become hardened, such as crispbread and cookies. The convenience market in Sweden is dominated by a handful of companies, some operated by retail companies primarily selling private label products. Only about 2% of the bread market in Sweden consists of home baking (Iakovlieva, 2021).

3.2. Quantification of the Swedish bakery supply chain

Two rounds of *stakeholder dialogues* (Sjölund et al., 2022; Mesiranta et al., 2023) were conducted with five industry actors operating within the Swedish TBA system, including two industrial bakeries, retailers, and logistic companies. The information shared enabled identification of current challenges and future potential within the bread supply chain, findings which were later used to quantify the surplus of bakery products sold under TBA and to develop scenarios representing a shift to alternative surplus pathways. Data disclosed by industry actors on surplus bakery products generated at bakery and retail level were aggregated, and the extrapolation variable used was market share based on sold products per year. The quantification of private-label bakery products included the same five major retailers as used in national statistics (Statistics Sweden, 2022) and values were extrapolated based on market share. Information on waste rates, sales, and annual production of private-label and bake-off bakery products was sourced via correspondence with bakeries and private actors, and supported with national data.

Through the stakeholder dialogues, along with literature, previous research, public company reports, and data shared via correspondence with industry actors and charity organizations, the volumes of surplus arising at the supplier-retailer interface were quantified. Alongside, the amount of surplus bread sold under TBA following different pathways was mapped. Loss rate records for bread and bakery products, monthly point-of-sale data, and surplus management data disclosed via correspondence with industry actors were used in waste quantification in this study. Additional data were collected from public reports (Polarbröd, 2020; Pågen, 2020; Fazer, 2022), previous research (Brancoli, 2021; Sjölund et al., 2023; Hildersten et al., 2025), and national statistics (Swedish Board of Agriculture, 2022). Based on annual consumption data, and accounting for waste occurring at different stages of the supply chain, *material flow analysis* (MFA) was used to map the level of production needed to support Swedish bakery goods consumption. Additional information on quantities is available via *Supplementary material* (Tables S1-S3). A second round of stakeholder dialogues was then conducted with relevant industry actors to verify the quantification and adjust the scenarios using their input.

Prevention pathways were defined in this study as measures that involve direct source reduction, while valorisation pathways were divided into high-value and low-value categories. High-value valorisation comprises measures for food redistribution, ultimately allowing surplus to be circulated back to the food system, such as food donations or price reductions. Low-value valorisation includes measures that repurpose the resource into other products, such as animal feed or ethanol used as fuel. Anaerobic digestion and incineration were considered in this study as pathways directed toward energy recovery.

3.3. Scenario development

The most promising pathways for reducing surplus and lowering environmental impacts that emerged during the stakeholder dialogues were used to formulate a total of nine tangible scenarios for bread management. The conventional *Base case* scenario was modeled to capture the current bread management pathways in Sweden. The results from the mapping of bakery supply chain in Sweden were used as input to design the pathways for surplus and waste in this scenario. Six alternative pathways were modeled as conceptual scenarios to simulate the impact of improvements applied either within the current system or without TBA in place, while the final two scenarios were developed to simulate optimal management. Fig. 1 presents an overview of all developed scenarios. To enable scenario analysis for each innovation separately, all scenarios were deliberately designed with minimal changes in parameters at a time.

The first scenario was identified through the stakeholder dialogues as an innovation applicable within the current TBA system that could reduce surplus bread, namely sharing data. In the absence of previous research on data-sharing specifically related to bread supplied under TBA, the reduction potential at retail and bakeries was calculated using

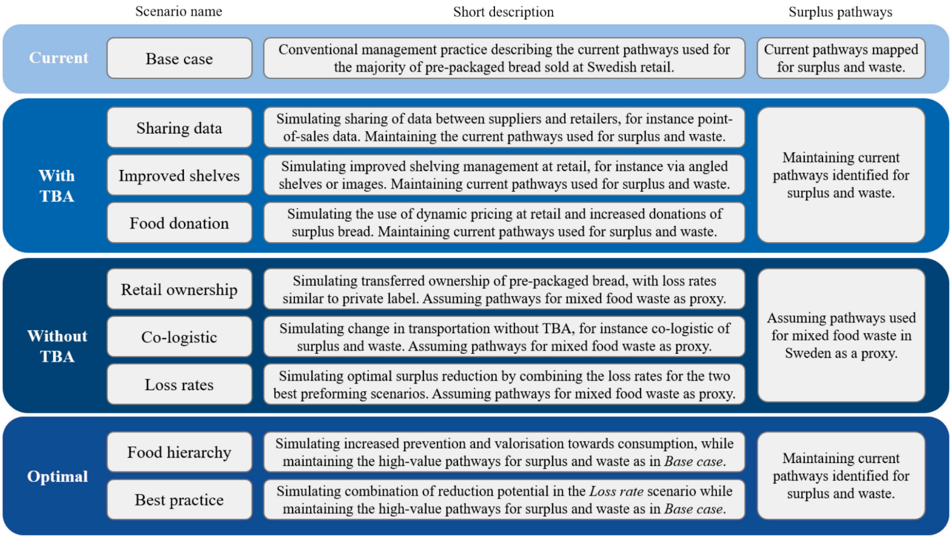

Fig. 1. Graphical illustration of the developed scenarios, specifying scenario name, short description, and surplus pathways. Included is the current management practice, six conceptual scenarios applied either within the current system or without a take-back agreement (TBA) in place, alongside two scenarios simulating optimal management.

data for milk. Nikolicic et al. (2021) found that 38% and 29% of waste was avoided for suppliers and retailers, respectively, when the TBA system was removed. These reduction factors were applied in the *Sharing data* scenario. Moreover, sub-optimal management of shelving together with a large product assortment was another risk factor for surplus bread generation at retail identified through the stakeholder dialogues. Previous studies have shown that by using angled shelves and images of bread, the feeling of abundance desired by consumers can be maintained while less bread needs to be placed on the shelves. A 50% reduction in surplus bread can be achieved at retail when incorporating images of bread in shelving units (Alm, 2021), while a 10-15% surplus can be avoided when using angled shelves (Easyfill, 2019). An average 31% reduction in the surplus generated at retail formed the modeling basis for the *Improved shelves* scenario. The final innovation applied within the TBA system was increased donations, since pre-packaged bread has a high potential to be redirected toward human consumption. Decentralised food donations were suggested as a pathway during the stakeholder dialogues, and the Swedish charity Sveriges stadsmissioner (2023) recently reported a 67% increase in demand for donated and price-reduced food. This scenario assumed that dynamic pricing could reduce surplus at retail by 21% (Sanders, 2024), favouring valorisation via food donations. The *Food donations* scenario assumed that all bread available via dynamic pricing was redirected toward human consumption via price reductions and donations.

A previously identified limitation of the TBA system is the low incentive for retailers to actively work toward reducing bread waste generated at their stores, since this bread is currently owned by the bakeries. If retailers were to take ownership of bread distributed under TBA, it is reasonable to assume that the loss rate would be similar to that of private-label bread already owned by retailers. A *Retail ownership* scenario was designed to simulate this shift, for which an average loss rate of 4.5% was calculated using data supplied by industry actors on waste rates for private-label bread. Waste transport distances required when removing TBA were used according to Weber et al. (2023) and the majority of the surplus was assumed to be directed toward anaerobic digestion and incineration (Johansson, 2021), since the lack of separate collection of bread would block pathways toward animal feed and ethanol production. The same surplus pathways, but applying the co-logistics distances suggested by Weber et al. (2023), were assumed in a *Co-logistics* scenario, which was requested by stakeholders and developed to simulate more streamlined transportation of bread as a way to reduce its climate impact. An optimal *Loss rate* scenario was developed to simulate the benefits of joint innovations at both bakery and retail level, by combining the loss rates used in *Sharing data* and *Retail ownership* scenario.

The final two scenarios were developed to simulate a theoretical optimal surplus management of bread, where the benefits of separate waste collection enabled via TBA are maintained but favouring more high-value pathways. A *Food hierarchy* scenario simulated the benefits of keeping TBA, but directing the surplus to human consumption via food donations and price reductions to a higher extent. A *Best practice* scenario also assumed that the TBA system was maintained, but applying the lowest waste rates at bakery and retail level. This scenario thus captured the benefits of reducing surplus while retaining the high-value pathways toward animal feed and ethanol production, which are otherwise often limited for mixed food waste.

3.4. Life cycle assessment

The environmental impacts of shifting from the conventional *Base case* scenario for surplus bread in Sweden to a conceptual scenario with or without TBA in place, was assessed using life cycle assessment following the ISO standards (ISO, 2006). A functional unit of *1 kg bread* sold to consumer at retail was selected to capture the scope of the study, following a cradle-to-gate approach. The system boundary accounted for inputs and outputs from ingredients, up to and including retail, alongside inputs needed for managing surplus following different pathways (Fig. 2). The software SimaPro 9.2 was used to model the scenarios, with datasets from Ecoinvent 3.8 and Agri-footprint 4.0. Although a shift from current to future scenarios is often described using consequential datasets, this study used cut-off datasets to ensure compatibility between the databases, since Agri-footprint implements Ecoinvent cut-off

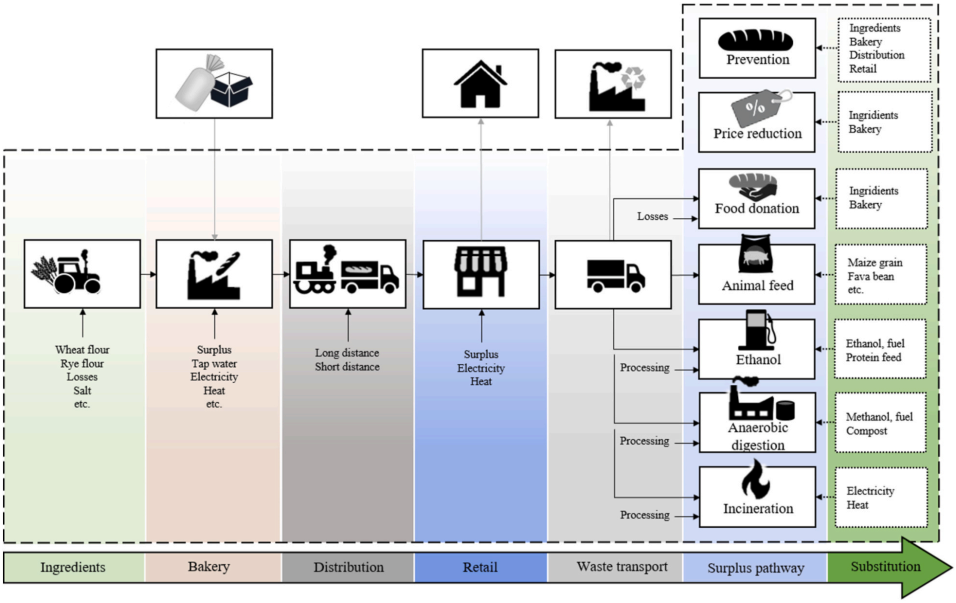

Fig. 2. Illustration of the modeled bread system in Sweden, indicating inputs and outputs accounted for along the value chain. The dashed line represents the system boundary, while the dotted line show the substitution included via system expansion. Outside the system boundary are production and recycling of packaging, alongside consumption and management of bread at households.

datasets for energy and fuel as background data (Blonk et al., 2022). Substitution via system expansion was used to account for avoided impact due to prevention and valorisation pathways for surplus bread. The impact of shifting from one scenario to another was calculated as the net difference between the two scenarios. The method Product Environmental Footprint (PEF) was used to assess environmental impact for 16 midpoint indicators, while the ReCiPe Endpoint (H) method was used to aggregate impact at midpoint to obtain a weighted result for three endpoint indicators. Supporting modeling inputs are available in Supplementary material (Tables S4-S12).

4. Results

4.1. Quantification of surplus bakery products

Based on the latest estimates of average consumption of baked goods (74.5 kg per person per year), annual consumption of bakery products in Sweden amounts to roughly 784 800 tonnes. Of this, industry-baked goods sold at retail level account for 79%, while roughly 11% originates from imports, 4% from small-scale production, including local bakeries and home baking, and 6% from the service sector, such as restaurants and schools. Of the baked goods sold at retail, 81% are pre-packaged products (Table 1). Our mapping also showed that 76% of all savoury bread is produced by industrial bakeries and sold under TBA, while the majority of sweet products (65%) are sold pre-packaged but distributed without TBA. Hardened products, both savoury and sweet, constitute 13% of retail products, while an additional 11% are non-packaged products.

At the national level, the quantification results showed that nearly 180 000 tonnes of baked goods (translating to roughly 17 kg per person) are wasted every year, with 51% of this waste originating from the supplier-retailer interface. Bakery products, both savoury and sweet,

Table 1
Quantity of different bakery products available at retail level in Sweden, expressed in tonnes annually.

Take-back agreement	Product category	Savoury	Sweet
Yes	Pre-packaged, soft	3.0×10^5	6.1×10^4
	Pre-packaged, hard	1.1×10^4	2.2×10^4
	Total	3.1×10^5	8.3×10^4
No	Pre-packaged, soft	1.6×10^4	1.6×10^5
	Pre-packaged, hard	2.3×10^4	4.8×10^4
	Total	3.9×10^4	2.1×10^5
No	Non-packaged, bake-off	4.7×10^4	1.6×10^4
	Non-packaged, convenience	1.4×10^4	1.4×10^4
	Total	6.1×10^4	3.0×10^4
	Total, retail level	**7.3×10^5**	

sold under TBA were found to be the largest surplus category at both the bakery and retail stages, while also being among the most wasted bakery products in households (Fig. 3).

4.2. Management pathways for surplus bread

Mapping of surplus pathways for bread sold under TBA along the supplier-retailer interface revealed that 86% of all bread produced is sold to consumers via retail (Fig. 4). The remaining 14% constitutes surplus bread not sold at retail (approximately 27 000 tonnes), with average loss rates of 6% at the bakery and 9% at retail levels. Less than 2% of the total surplus is currently redirected toward human consumption via reduced prices or food donations, while the majority is instead directed toward energy recovery (59%), ethanol production (22%), and animal feed (17%).

All alternative scenarios applied to the Swedish bread system, with or without TBA in place, resulted in an overall reduction in surplus bread. The change in waste rates when shifting from the conventional

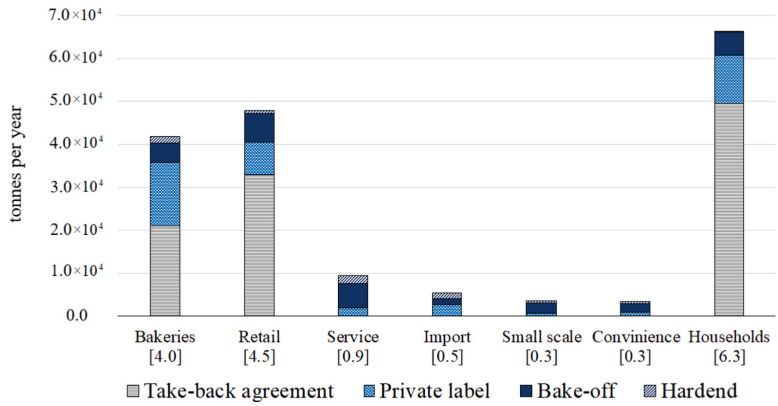

Fig. 3. Quantification of surplus bakery products generated annually along the value chain, including both savoury and sweet products. Numbers in brackets represent the amount of surplus in kg per person and year.

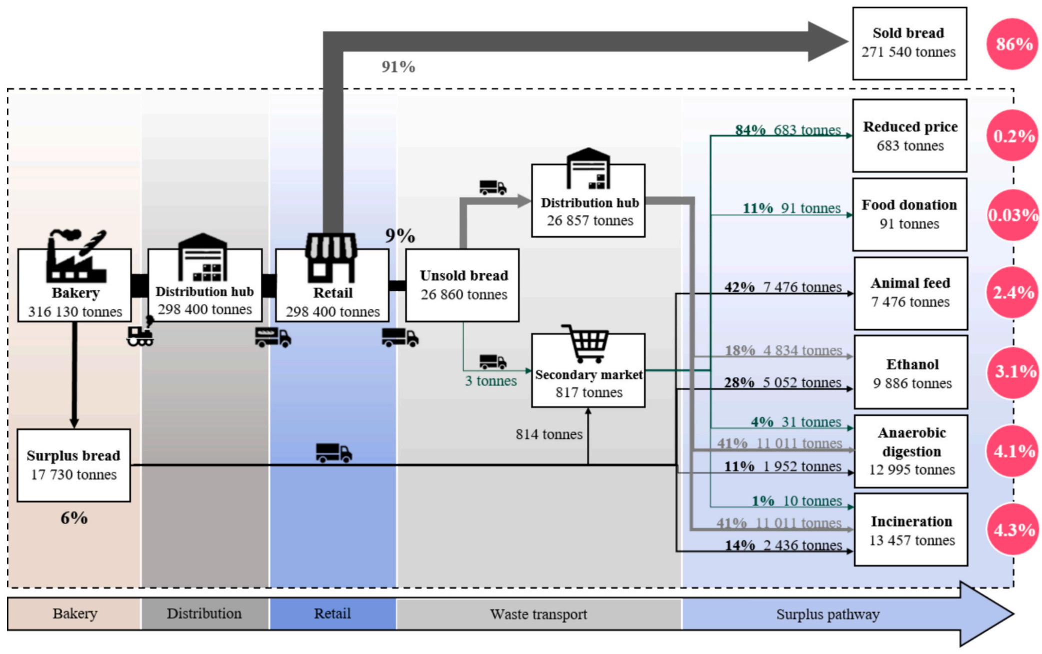

Fig. 4. Mapping of the current *Base case* scenario, indicating loss rates and management pathways, for surplus and waste generated at different stages of the supplier-retailer interface. The distribution of pathways (%) is shown per kg of produced bread.

Base case scenario is shown in Table 2, along with the corresponding waste reduction potential and alternative pathways for surplus products at the bakery and retail levels. The highest waste prevention potential was found for the *Sharing data* and *Retail ownership* scenarios, and the scenarios assuming their loss rates. Both the *Best practice* and the *Loss rates* scenario resulted in 20 000 tonnes less bread wasted at the supplier-retailer interface, while the *Food donation* and *Food hierarchy* scenarios yielded the highest share of valorisation toward human consumption.

4.3. Environmental impacts of bread scenarios

In terms of climate impact and damage to ecosystems, the conventional *Base case* scenario was found to result in 1.0 kg CO_2eq and loss of 3.9×10^{-8} species.year per kg bread. The results further show that the impact at midpoint and damage at endpoint decreased for the majority

Table 2
Loss rate and surplus quantities not sold via the intended pathway, in relation to amount produced, for each of the nine scenarios assessed.

	Base case	Sharing data	Improved shelves	Food donation	Retail ownership	Co-logistics	Loss rates	Food hierarchy	Best practice
Loss rate (%)									
Bakery	6.0%	3.4%	6.0%	6.0%	6.0%	6.0%	3.4%	6.0%	3.4%
Retail	9.0%	6.4%	6.2%	7.1%	4.5%	9.0%	4.5%	9.0%	4.5%
Surplus (tonnes)									
Bakery	1.8×10^{4}	1.1×10^{4}	1.8×10^{4}	1.8×10^{4}	1.8×10^{4}	1.8×10^{4}	1.1×10^{4}	1.8×10^{4}	1.1×10^{4}
Retail	2.7×10^{4}	1.9×10^{4}	1.9×10^{4}	2.1×10^{4}	1.3×10^{4}	2.7×10^{4}	1.3×10^{4}	2.7×10^{4}	1.3×10^{4}
Pathway (tonnes)									
Prevention	0	1.4×10^{4}	8.3×10^{3}	0	1.3×10^{4}	0	2.0×10^{4}	0	2.0×10^{4}
Reduced price	6.8×10^{2}	4.5×10^{2}	5.6×10^{2}	5.6×10^{3}	8.8×10^{2}	1.3×10^{3}	7.0×10^{2}	1.3×10^{4}	3.8×10^{2}
Donations	9.1×10^{1}	7.0×10^{1}	7.4×10^{1}	1.5×10^{3}	2.9×10^{2}	4.2×10^{4}	2.3×10^{2}	1.3×10^{4}	5.1×10^{1}
Animal feed	7.5×10^{3}	4.8×10^{3}	6.1×10^{3}	6.5×10^{3}	0	0	4.2×10^{3}	8.9×10^{3}	4.2×10^{3}
Ethanol	9.9×10^{3}	6.7×10^{3}	8.0×10^{3}	6.2×10^{3}	0	0	2.8×10^{3}	4.5×10^{3}	5.5×10^{3}
Anaerobic digestion	1.3×10^{4}	9.2×10^{3}	1.1×10^{4}	9.4×10^{3}	1.0×10^{4}	1.3×10^{4}	6.7×10^{3}	2.2×10^{3}	7.2×10^{3}
Incineration	1.3×10^{4}	9.5×10^{3}	1.1×10^{4}	1.1×10^{4}	2.0×10^{4}	3.0×10^{4}	1.0×10^{4}	2.2×10^{3}	7.5×10^{3}

of alternative scenarios when simulating a shift from the current system (Table 3). Shifting from the *Base case* scenario to any of the alternative scenarios was found to reduce the climate impact. In general, the *Food hierarchy* and *Best practice* scenario returned the highest environmental savings per kg bread at both midpoint and endpoint level, while shifting to the *Co-logistics* scenario resulted in an increased impact for 12 of the 16 midpoint categories and two of three endpoint categories.

With respect to climate, the annual impact for the current *Base case* scenario was translated to roughly 46 000 tonnes CO_2eq (Fig. 5). The primary impact hotspot for all scenarios was production of ingredients, primarily wheat and rye cultivation (see Fig. S14 in *Supplementary material*), while retail operations and bakery processing contributed very little to impact at midpoint level. Substitution, constituting prevention and valorisation pathways, accounted for via system expansion was also found to greatly influence the overall result for most scenarios.

The results further show that scenarios with a high use of prevention and high-value valorisation resulted in larger environmental savings, while anaerobic digestion and incineration resulted in five-fold and 10-fold lower climate benefits, respectively (Table 4). Environmental impact for all assessed midpoint and endpoint categories are available in *Supplementary material* (Table S15).

5. Discussion

Mapping the Swedish bakery sector, accounting for surpluses and losses of both savoury and sweet products along the supply chain, is one of the key outcomes of this study. We found that roughly 784 800 tonnes of bakery products are consumed annually in Sweden, of which 51% are sold and distributed under TBA. Approximately 14% of the savoury bread distributed under TBA is never sold at retail, due to the surplus generated along the supplier-retailer interface. This is in line with the loss rates for bread of 10-13% of production volume estimated by

Table 3
Environmental impacts per kg bread from the *Base case* scenario, alongside the impact per functional unit of shifting from the current *Base case* scenario for each alternative scenario. Green: reduced impact. Gray: increased impact.

Midpoint level	Base case	Sharing data	Improved shelves	Food donation	Retail ownership	Co-logistics	Loss rates	Food hierarchy	Best practice
Climate change	1.0×10^{0}	-6.4×10^{-2}	-2.8×10^{-2}	-1.3×10^{-2}	-5.7×10^{-2}	-1.8×10^{-2}	-7.7×10^{-2}	-7.4×10^{-2}	-9.0×10^{-2}
Ozone depletion	1.2×10^{-7}	-9.9×10^{-9}	-4.9×10^{-9}	-2.4×10^{-9}	-1.1×10^{-8}	-4.8×10^{-9}	-1.3×10^{-8}	-1.4×10^{-8}	-1.6×10^{-8}
Ionising radiation	3.5×10^{-1}	-1.1×10^{-2}	-2.8×10^{-3}	-2.3×10^{-3}	-1.8×10^{-2}	-2.2×10^{-2}	-2.4×10^{-2}	-4.0×10^{-3}	-1.2×10^{-2}
Ozone formation	5.9×10^{-2}	-2.7×10^{-3}	4.4×10^{-5}	-5.3×10^{-6}	1.3×10^{-3}	3.9×10^{-3}	-1.6×10^{-4}	-3.3×10^{-3}	-3.0×10^{-3}
Particulate matter	1.9×10^{-7}	-1.1×10^{-8}	-2.9×10^{-9}	-1.7×10^{-9}	-4.0×10^{-9}	3.8×10^{-9}	-8.3×10^{-9}	-1.4×10^{-8}	-1.4×10^{-8}
Human toxic $_{\text{non-cancer}}$	4.5×10^{-8}	-2.8×10^{-9}	-1.2×10^{-9}	-7.5×10^{-10}	-1.5×10^{-9}	4.3×10^{-10}	-2.5×10^{-9}	-3.8×10^{-9}	-3.0×10^{-9}
Human toxic $_{\text{cancer}}$	1.4×10^{-9}	-8.4×10^{-11}	-3.6×10^{-11}	-2.1×10^{-11}	-3.6×10^{-11}	2.9×10^{-11}	-6.7×10^{-11}	-1.1×10^{-10}	-9.3×10^{-11}
Acidification	5.3×10^{-2}	-2.6×10^{-3}	-2.0×10^{-4}	-1.6×10^{-4}	7.3×10^{-4}	3.0×10^{-3}	-5.8×10^{-4}	-3.2×10^{-3}	-2.8×10^{-3}
Eutrophication $_{\text{fresh}}$	4.2×10^{-4}	-2.6×10^{-5}	-1.0×10^{-5}	-6.7×10^{-6}	-1.4×10^{-5}	2.4×10^{-6}	-2.4×10^{-5}	-3.3×10^{-5}	-2.7×10^{-5}
Eutrophication $_{\text{marine}}$	3.0×10^{-2}	-1.5×10^{-3}	-1.8×10^{-4}	-1.4×10^{-4}	3.0×10^{-4}	1.6×10^{-3}	-4.6×10^{-4}	-1.9×10^{-3}	-1.6×10^{-3}
Eutrophication $_{\text{terrestrial}}$	2.8×10^{-1}	-1.3×10^{-2}	-6.0×10^{-4}	-5.9×10^{-4}	5.0×10^{-3}	1.7×10^{-2}	-2.2×10^{-3}	-1.7×10^{-2}	-1.4×10^{-2}
Ecotoxicity $_{\text{freshwater}}$	8.2×10^{1}	-4.9×10^{0}	-1.9×10^{0}	-1.2×10^{0}	-2.1×10^{0}	1.3×10^{0}	-4.0×10^{0}	-6.7×10^{0}	-5.3×10^{0}
Land use	1.1×10^{2}	-7.2×10^{0}	-3.1×10^{0}	-2.1×10^{0}	-4.4×10^{0}	2.8×10^{-1}	-7.1×10^{0}	-9.9×10^{0}	-7.6×10^{0}
Water use	6.1×10^{0}	-3.8×10^{-1}	-1.8×10^{-1}	-1.2×10^{-1}	-2.1×10^{-1}	4.1×10^{-1}	-3.6×10^{-1}	-5.6×10^{-1}	-4.0×10^{-1}
Resource use $_{\text{fossils}}$	1.4×10^{1}	-8.9×10^{-1}	-3.8×10^{-1}	-2.0×10^{-1}	-9.4×10^{-1}	-5.1×10^{-1}	-1.2×10^{0}	-1.1×10^{0}	-1.3×10^{0}
Resource use $_{\text{mineral}}$	9.1×10^{-6}	-4.5×10^{-7}	-1.3×10^{-7}	-5.9×10^{-8}	-9.1×10^{-8}	2.6×10^{-7}	-2.9×10^{-7}	-4.6×10^{-7}	-5.1×10^{-7}
Endpoint level									
Human health	6.7×10^{-6}	-3.5×10^{-7}	-6.5×10^{-8}	-3.8×10^{-8}	-1.8×10^{-8}	2.7×10^{-7}	-1.2×10^{-7}	-1.2×10^{-7}	-4.1×10^{-7}
Ecosystems	3.9×10^{-8}	-2.3×10^{-9}	-7.2×10^{-10}	-4.8×10^{-10}	-7.9×10^{-10}	8.4×10^{-10}	-7.5×10^{-10}	-7.5×10^{-10}	-2.4×10^{-9}
Resources	8.8×10^{-2}	-7.1×10^{-3}	-3.4×10^{-3}	-1.7×10^{-3}	-7.2×10^{-3}	-2.8×10^{-3}	-3.5×10^{-3}	-3.5×10^{-3}	-1.1×10^{-2}

Climate change: kg CO_2eq, ozone depletion: kg CFC_{11}eq, ionising radiation: kBq U-235eq, ozone formation: kg NMVOCeq, particulate matter: disease inc., human toxic: CTUh, acidification: mol H^+eq, Eutrophication$_{\text{fresh}}$: kg Peq, Eutrophication$_{\text{marine}}$: kg Neq, Eutrophication$_{\text{terrestrial}}$: mol Neq, Ecotoxicity$_{\text{freshwater}}$: CTUe, land use: Pt, water use: m^3 depriv., Resource use$_{\text{fossils}}$: MJ, Resource use$_{\text{mineral}}$: kg Sbeq. Human health: DALY, ecosystems: species·year, resources: USD2013.

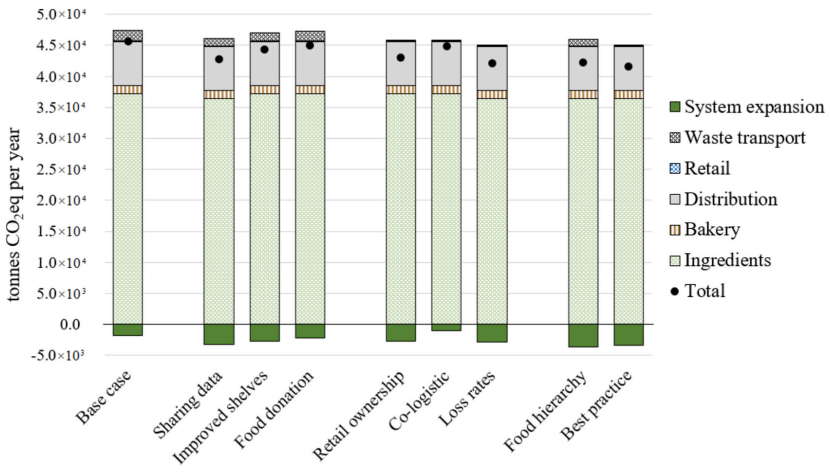

Fig. 5. Climate impact of bread on a national level, indicating each process contribution to total impact for all nine scenarios. Negative values indicate avoided impact due to system expansion via substitution of products from surplus bread pathways.

Table 4
Climate impact and ecosystem damage per kg of bread following each surplus pathway.

	Prevention	Reduced price	Donation	Animal feed	Ethanol	Anaerobic digestion	Incineration
kg CO_2eq	-1.0×10^{0}	-8.4×10^{-1}	-5.2×10^{-1}	-4.8×10^{-1}	-4.3×10^{-1}	-1.8×10^{-1}	-1.0×10^{-1}
Species.year	-3.8×10^{-8}	-3.8×10^{-8}	-2.4×10^{-8}	-9.2×10^{-9}	-2.9×10^{-8}	-4.1×10^{-10}	-7.7×10^{-10}

Goryńska-Goldmann et al. (2021b) and the bread return rates of 4-17% in Germany reported by Ritter et al. (2015). However, it is considerably higher than the 5% rate of surplus bread generation reported for small-scale bakeries in an Italian study (Pietrangeli et al., 2023).

The Swedish loss rate translates to annual wastage of around 27 000 tonnes of still edible bread sold under TBA, making it the single largest category of bakery surplus (54%) generated at retail (Fig. 3). Brancoli et al. (2019) also identified TBA bread as the largest waste category at retail (55%), but estimated a slightly lower quantity of 15 500 tonnes wasted bread. A plausible explanation for the similar waste fraction but higher quantities found in this study is the increased consumption of bread over the years, while the TBA system has remained unchanged. Using the price of a "*Lingongrova*" loaf (25 SEK per 500 g), marketed as "*Sweden's most purchased bread*" (Pågen, 2024), this corresponds to an economic loss of around 1.4 billion SEK every year only at retail. Applying the same price per 500 g to estimate the economic value of all bakery products wasted, roughly 9 billion SEK (780 million EUR) are lost via surplus each year. Despite the relatively low price of bread in Sweden, the vast quantities of surplus add up to a considerable amount of economic losses, in addition to the social and environmental consequences of under-utilising food resources.

Although surplus bread is often of good quality and suitable for high-value valorisation, our results show that the largest surplus pathway is currently anaerobic digestion, followed by incineration, and ethanol production (Table 2). This supports previous findings on food waste pathways in Sweden by Johansson (2021). Only 2% of surplus bread sold under TBA is directed toward human consumption via reduced prices or food donations (Fig. 3), likely since this pathway currently requires incentives in terms of time and money to facilitate. Of the bread directed toward human consumption, our mapping showed that 5% is currently diverted to anaerobic digestion or incineration. This is in line with findings for food donations by Sundin et al. (2023), and emphasises an important limitation related to efficiency and management in current high-value valorisation pathways for surplus food. These results confirm that the existing TBA system can indeed be considered a cause of bread waste at both supplier and retail level. More importantly, the current TBA system tends to limit use of high-value pathways for unsold products, since energy production is favoured over human consumption. Therefore, one could argue that the bread system would greatly benefit from innovations that promotes pathways for prevention and valorisation.

The potential benefits of shifting from the current system, or applying changes to it, were assessed using different scenarios. *Sharing data* between suppliers and retailers operating within the conventional TBA system, or implementing *Retail ownership* of all bread sold at retail without TBA in place, reduced surplus by 30% (Table 2). At national level, this represents a reduction in surplus of over 13 000 tonnes annually. The threshold to share data should arguably be quite low, since much of the data are already being collected, but sharing agreements between supplier and retailer will need to be put in place for this solution to become reality. However, since businesses often prioritise economic aspects rather than the environmental impacts of their operations, there can be considerable barriers or conflicts of interest preventing them from actually sharing data, as also noted by Winkler et al. (2023). If the TBA system were instead to be completely removed and replaced with *Retail ownership*, emulating the current system for private-label bread, less surplus would be generated at retail. The transfer of ownership to retail would not necessarily affect the pathways used for any bread waste generated, but would rather support further prevention measures, like coordination of promotions in the store and discounts at the end of shelf life. Although this assumption is plausible, it is also important to highlight the potential burden shifting related to actions targeted at reducing waste at retail. For instance, reducing prices at retail might lead to increased bread waste at households when consumers are enticed to buy more than will be consumed. This is especially relevant to consider for price reduction pathways, alongside, for instance, bake-off products with a shorter shelf life compared to prepackaged bread. However, allowing retailers to influence the management of bread sold in their stores would likely reduce surplus generation via prevention.

Even though the *Sharing data* and *Retail ownership* scenarios involved similar loss rates, the former focused more on waste prevention actions, while the later both directed more toward human consumption while also impeding pathways toward animal feed and ethanol production. The *Best practice scenario* showed that there are considerable climate and ecosystem benefits of keeping the current surplus pathways enabled via TBA, but at the same time allowing retailers to influence the management of bread. When considering all impact categories at midpoint, shifting to the *Food hierarchy* scenario and ultimately using food resources according to their highest value enabled the highest environmental benefits. Although the climate impact was reduced when simulating a shift to *Co-logistics* scenario, the shift also inferred increased impact for a majority of other categories, including acidification, eutrophication, land use, and ecosystem damage (Table 3). This emphasizes the importance of including a broad range of impact categories, as crucial aspects might otherwise be overlooked.

Furthermore, it is important to note that there are other potential trade-offs between the scenarios. For instance, if the pathways for prevention, price reduction, and food donations are opened up, then less fossil fuel can be substituted, as less ethanol and biogas would be produced. Increased donations are both required and feasible within the current *Base case* scenario, but would require adaptation and agreements for distributing the bread to people in need. The resource requirements of food aid organisations, which often operate on a non-profit basis, are also a significant factor for the feasibility of these scenarios (Mesiranta et al., 2022). Favouring decentralised donations, for example, by allowing people to pick up bread directly at retail, could reduce the need for additional management outside the retail stage.

Modeling the innovations as separate scenarios allowed evaluation of the individual actions, but many of the innovations assessed via scenarios can, and arguably should, be combined with other non-overlapping actions to enhance the positive effects. For instance, *Food donation* could be combined with *Sharing data*, *Retail ownership* or *Improved shelves* to enhance the positive effects of prevention and circulating bread back to the food system. A similar rationale applies to the *Co-logistics* scenario, which in itself did not contribute to reduced surplus, but rather focused on optimising transport and management, which can be a limitation for other scenarios. The *Co-logistics* scenario could in theory be combined with any other scenario assessed, and would benefit pathways that would otherwise require increased transportation, such as the *Food donation* scenario. The power of prevention and valorisation pathways was even more evident when each pathway was analysed separately (Table 4), where prevention gave five-fold and 10-fold higher climate savings than anaerobic digestion and incineration, respectively. These benefits will likely be maintained when prevention is favoured, as illustrated by the *Food hierarchy* scenario.

5.1. Environmental benefits of prevention and valorisation

The conventional *Base case* scenario was found to contribute an annual climate impact of nearly 46 000 tonnes CO_2eq and an ecosystem damage of close to 2 species per year at national level (Table 3). Keeping in mind that factors such as type of bread, energy use, and LCA methodology heavily influence the environmental performance of bread, the climate impact of 1.0 kg CO_2eq per kg bread sold under TBA found in this study follows the same order of magnitude as reported in previous studies. When assessing the impact of rye bread produced in Sweden, Hildersten et al. (2025) found that 1 kg of bread resulted in 0.81 kg CO_2eq. This is in line with the present results, although their study also included packaging, which was omitted in this study. When assessing the carbon footprint of bread in the United Kingdom, Espinoza-Orias et al. (2011) found a climate impact ranging from 1.2 to 1.5 kg CO_2eq per kg of bread, which was similar to the range reported by Kulak et al. (2015) of 0.6 to 1.7 kg CO_2eq per kg of bread. The differences in climate impact per kg bread could be a consequence of different methodological choices, including system boundaries and how the studies dealt with multifunctionality. When accounting for the climate benefits of prevention and valorisation of surplus bread at the supplier-retailer level in Norway, Svanes et al. (2019) found a climate impact of 1 kg CO_2eq per kg bread. However, although the impact per functional unit was similar to that in our study, their study included, for example, more transportation of ingredients and omitted valorisation pathways toward human consumption. Important to note is the considerable range in climate impact of bread reported in previous research, also highlighted by Notarnicola et al. (2017) and later by Rayichuk et al. (2023). Accounting for the avoided environmental impacts of substituted ingredients in LCA studies could aid in determining the valorisation pathway with the best environmental outcome, a conclusion also reached by Thorsen et al. (2024). The result per kg bread following a prevention or valorisation pathway in this study (Table 4) is consistent with previous findings by Brancoli et al. (2020) using a similar modeling approach. However, in the present study, we incorporated updated values from Ecoinvent, literature and input obtained via stakeholder dialogues, which explains the slight difference in numerical values. Another important aspect is that the present study combined data from both Ecoinvent and Agri-footprint, as this enabled a better description of the inputs used. To ensure compatibility between the databases, this also inferred that the study used cut-off as allocation principle for the inputs and outputs. Since LCA methodology heavily influence the results, this should be accounted for when evaluating the result.

Production of ingredients and distribution of bread from bakery to retail were found to be the primary hotspots with respect to climate impact, as expected due to the high dependency on fossil fuels during production and transport. This is in line with previous findings on the climate impact of bread (Svanes et al., 2019; Nadi et al., 2022; Weber et al., 2023). It should also be highlighted that our results show that prevention and valorisation to human consumption provide the greatest climate and ecosystem benefits, since these pathways reduce the demand for new materials. Although ecosystem damage has been identified as an increasingly important aspect to consider, especially regarding food systems, this method is rarely adequately included in LCA research (Gabel et al., 2016). To our knowledge, no previous study has accounted for the ecosystem damage related to the bread system, making the outcome from this study unique. The results can be used as a foundation for future comparison and validation, while the numerical value for ecosystem damage should primarily serve as an indicator for damage hotspots.

5.2. Market power for reduced bread waste

The use of TBA has often been attributed to the high market power of Swedish retailers, but similar agreements are used in multiple countries (Brancoli et al., 2019). The system ensures that bread waste is collected separately from other food waste fractions, opening up for alternative pathways for waste management and valorisation of unsold bread. Ideally, all bread produced should be eaten, either directly or following conversion into new food products. Incorporating environmental aspects into business management has been shown to positively influence consumer attitudes, benefit retailer branding, and support further implementation of high-value food recovery. As the success and efficiency of food retailers are also linked to corporate social responsibility (Kulikovskaja and Aschemann-Witzel, 2017), food retailers can reap multiple benefits from actions to reduce food waste without compromising customer satisfaction or marketability.

Considering that the Swedish bread supply chain is dominated by three industry bakeries, it is nonetheless important to question the potential to further reduce the surplus generated at the supplier-retailer interface. The Swedish bread system may already be quite optimized, as we observed only a 14% loss from production to retail for TBA bread. Further prevention measures might not be economically feasible for bakeries or retailers, which is another aspect that should be evaluated. Although some pathways favoured for redirecting surplus bakery products may not result in less overall surplus being generated, they can still contribute to higher value recovery of available resources. The lowest waste rates were found for hardened bakery products, both savoury and sweet, due to their considerably longer shelf-life. Therefore, a higher consumption share of hard bread would likely reduce overall bread waste. Consuming more products from these categories could also become an important aspect of food preparedness and self-sufficiency. Hard savoury bread can often be considered healthier than soft bread, containing more whole grains and fibre. Thus, shifting to a higher share of hard bread for the purposes of waste reduction could also bring health benefits (Edgar et al., 2022). Introducing more bread of any kind into the human diet, preferably at the expense of animal-based foods, could also reduce environmental burdens related to a nutritionally balanced diet (Kramer et al., 2018).

While the TBA as a business model requires separate collection of bread, it has also been identified as a risk factor for bread waste generation (Eriksson et al., 2017). The positive correlation between TBA and high levels of waste can be explained by the constant battle between suppliers and retailers, and among suppliers themselves, over the service level on the shelves. Thus, while TBA contributes to the circularity of the bread supply chain, it also has the potential to increase waste generation, presenting a complex challenge for sustainability in the industry. In April 2019, EU Directive 2019/633 categorized TBA as potentially unfair trading practice, which has made this business model less frequent in other countries (Pietrangeli et al., 2023). The TBA model is not enforced by any policy or regulation, and bakeries are in theory free to use any business model they choose. However, as shown by Ghosh and Eriksson (2019), the TBA has become the trade standard in Sweden, and bakeries therefore often feel pressured to adopt this business model. The results from this study show that the current TBA model generates most surplus and performs worst with respect to almost all midpoint and endpoint categories assessed, while considerable savings in both waste generation and environmental impact were found for all alternative scenarios. Since many of these scenarios could be implemented into the current bread supply chain, either directly or following minor modifications, it should be emphasized that taking any action toward prevention and valorisation of surplus bread is more important than optimizing the actions taken. Given that the current TBA system is already in place and works well in many aspects, it can be argued that innovations to improve this system should be prioritized initially. Identifying the benefits and favouring pathways that allow high-value recovery of surplus bakery products can enable a more purposeful use of resources, ensuring that the food we produce is consumed by humans rather then used feed the energy sector. In turn, this supports efficient food systems that will benefit the environment, society, and the economy.

5.3. Limitations and future outlook

Although quantification via self-assessment tends to yield underestimates, this method engaged stakeholders and provided important insights into bread pathways. Direct measures of surplus or waste were not possible in this study, as most production data are currently not openly shared by bakeries or retailers. In waste quantification, we instead used inputs from literature, previous research, and information shared in stakeholder dialogues, alongside data provided by industry actors. We attempted to validate the production quantities, loss rates, and surplus pathways for bread with multiple industry actors, and the results were accepted by two independent actors operating within the Swedish bakery system. The in-depth mapping mainly focused on savoury bread distributed under TBA, while the quantification of the remaining bakery products was only addressed at national level. To capture the environmental impact of the entire bakery sector, the life cycle of sweet and hardened products should also be assessed. Future studies would also benefit from including multiple impact categories to allow in-depth analysis of environmental impact, which is especially important when evaluating the performance of food systems.

Alongside Agenda 2030 and SDGs, the Swedish government has set a goal to reduce food waste by 20% until 2025, and to increase the amount of food entering the retail and consumer level. Retailers can play a vital role in this food waste reduction effort, as they have a unique opportunity to influence the prevention of surplus bread generation at all stages of the value chain. However, we need to overcome existing barriers to allow new pathways to thrive. The current approach to managing surplus bread in Sweden is problematic, as energy production (e.g. ethanol and biogas production) is currently prioritized over prevention and valorisation. The solution to the food waste issue is thereby largely framed as an efficiency problem, focusing on optimizing material flows and avoiding waste, rather than as an issue of solidarity, where already produced food is primarily redirected toward human nutrition (Mesiranta et al., 2022).

To support sustainable production and consumption of food in the future, it is crucial to prevent losses and valorise surpluses for human consumption. Producing any kind of food and using it to feed animals or for energy production should not be considered sustainable from an environmental, social, or economic point of view. On the contrary, we need strong incentives to drive the required change toward sustainable food systems, where production and consumption are in balance, and food is used to its fullest potential. Even though a system or process might be considered circular, it can still contribute to reduced sustainability or increased environmental impact if the resources required to maintain circularity outweigh the benefits. This highlights an important goal conflict between circularity and resource use, which is especially critical for food systems with high potential for increased circularity. If these goal conflicts are not recognized and adequately addressed, even well-intentioned innovations and incentives could risk contributing to reduced sustainability or increased environmental impact. Since the environmental benefits of prevention and valorisation of surplus bread were found to be offset by the climate cost of managing the bread, this aspect is also relevant to consider for bread supply chains. Therefore, any action or innovation applied to the bakery supply chain to reduce waste generation must be feasible, yet adequate. Stakeholder dialogue, modeling scenarios, policy recommendations, and industry collaboration can all be valuable tools for identifying and implementing changes to the bread supply chain, without jeopardizing profitability, development, or consumer satisfaction.

6. Conclusions

Two key outcomes of this study are, firstly, the quantification of surplus bread and bakery products generated at different stages of the supply chain, including both sweet and savoury products, and secondly the identification of current pathways used to manage surplus. The results show that just under 180 000 tonnes of baked goods, equivalent to roughly 780 million EUR, are lost or wasted annually. The majority of savoury bread consumed in Sweden was found to be sold and distributed under TBA, where 14% of production (27 000 tonnes) becomes surplus already at the supplier-retailer interface each year. The results further show that the loss rate for sweet products is overall higher compared to savoury products, with even lower loss rates for hardened products. The power of prevention and valorisation was demonstrated through scenario analyses, where innovations for waste reduction and alternative surplus pathways at the retail-bakery interface were simulated. The largest reduction potential was obtained for preventative actions, either through sharing of data or price reductions. Scenarios adopting both prevention and high-value valorisation resulted in up to ten times lower climate impact per kg bread. The outcome of this study can be directly used to support industry actors who want to implement changes that reduce waste or promote high-value valorisation pathways. The results can also provide guidance in developing policy recommendations that economically favour prevention and valorisation toward human consumption, and provide a valuable basis for future research on resource-efficient food systems.

CRediT authorship contribution statement

L. Bartek: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **A. Sjölund:** Writing – review & editing, Methodology, Investigation, Data curation. **P. Brancoli:** Writing – review & editing, Writing – original draft, Validation, Supervision, Software, Methodology, Formal analysis, Data curation. **C. Cicatiello:** Writing – review & editing, Validation, Formal analysis. **N. Mesiranta:** Writing – review & editing, Validation, Formal analysis. **E. Närvänen:** Writing – review & editing, Validation, Formal analysis. **S. Scherhaufer:** Writing – review & editing, Validation, Formal analysis. **I. Strid:** Writing – review & editing, Supervision, Methodology. **M. Eriksson:** Writing – review & editing, Supervision, Project administration, Methodology, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This work was supported by the H2020 project LOWINFOOD (Multi-actor design of low-waste food value chains through the demonstration of innovative solutions to reduce food loss and waste). LOWINFOOD is funded by the European Union's Horizon 2020 - Research and Innovation Framework Programme under Grant Agreement no. 101000439. The views reflected in this article represent the professional views of the authors and do not necessarily reflect the views of the European Commission or other LOWINFOOD project partners.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.spc.2025.01.013.

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Acta Universitatis Agriculturae Sueciae

Doctoral Thesis No. 2025:89

Large amounts of food are lost and wasted along the supply chain, while potentially valuable resources remain untapped. This thesis examines how, and under what conditions, recovery pathways designed to reintegrate surplus food and by-products into the food system can support resource efficiency and reduce environmental impacts in food systems. The results show that strategically targeted recovery, guided by the principle of using resources at their highest possible value and with high substitution effects, can yield substantial environmental benefits. In doing so, these pathways could transform food once considered waste into key resources for a sustainable food future.

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ISSN 1652-6880
ISBN (print version) 978-91-8124-073-3
ISBN (electronic version) 978-91-8124-119-8