Acta Universitatis Agriculturae Sueciae

Doctoral Thesis No. 2026:28
Faculty of Natural Resources and Agricultural Sciences

Climate impacts from beef and dairy production in Sweden

Mitigation potential and multifunctionality

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SWEDISH UNIVERSITY OF AGRICULTURAL SCIENCES

DOCTORAL THESIS
Uppsala 2026

Acta Universitatis Agriculturae Sueciae
2026:28

ISSN 1652-6880
ISBN (print version) 978-91-8124-245-4
ISBN (electronic version) 978-91-8124-275-1
https://doi.org/10.54612/a.4ckgguot8o

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Print: SLU Grafisk service, Uppsala 2026

Climate impacts from beef and dairy production in Sweden - mitigation potential and multifunctionality

Abstract

Cattle production contributes substantially to greenhouse gas emissions but also provides important functions beyond food production, such as ecosystem services from grazing, which may be overlooked in climate mitigation. This thesis increases knowledge on how climate impacts from beef and milk production can be quantified and reduced while considering multifunctionality and trade-offs. It investigates a method to include ecosystem services in climate impact assessments, evaluates feed-related mitigation strategies, and explores scenarios for expanding grazing on semi-natural pastures under methane limits. Including non-provisioning ecosystem services in climate impact assessments of beef and milk can have a major effect: up to 48% of emissions were attributed to these services rather than beef or milk, helping reduce the risk of losing multifunctionality in climate mitigation efforts. Feeding suckler cows cereal straw and grass-clover silage in winter did not reduce climate impact of beef due to soil carbon and deforestation emissions, but could increase food production and maintenance of semi-natural pastures, supporting ecosystem services. Giving 3-NOP to dairy cows reduced milk's climate impact by 12% on an intensive farm (from 0.74 to 0.65 kg CO_2e/kg milk) and by 9% on an extensive farm (from 0.99 to 0.91 kg CO_2e/kg milk). While mitigation potential was higher on the intensive farm, the extensive farm provided more co-benefits, such as ecosystem services from grazing semi-natural pastures. Scenarios exploring ecosystem service delivery and biodiversity conservation through increased semi-natural pasture management showed that adjustments in current livestock systems (e.g., castrating bulls and keeping them as low-intensity grazing steers) could expand grazed semi-natural pastures in Sweden without increasing methane emissions. Accepting a 10% methane increase could double pasture areas. Overall, this thesis shows how climate impacts from Swedish beef and dairy production can be assessed and reduced while considering multifunctionality, highlighting the need for a systems perspective integrating emissions, ecosystem services, and food production.

Keywords: cattle, life cycle assessment, ecosystem services, semi-natural pastures, alternative feeds, feed additives, sustainable food systems, land use

Klimatpåverkan från svensk nöttkötts- och mjölkproduktion – reduktionspotential och multifunktionalitet

Abstract

Nötkreatursproduktion bidrar avsevärt till växthusgasutsläpp, men även till viktiga funktioner utöver matproduktion, såsom ekosystemtjänster från bete. Dessa riskerar att förbises i klimatåtgärder. Denna avhandling bidrar till ökad kunskap om hur klimatpåverkan från nötkötts- och mjölkproduktion kan beräknas och reduceras, samtidigt som multifunktionalitet och målkonflikter beaktas. Avhandlingen undersöker en metod för att inkludera ekosystemtjänster i klimatberäkningar, utvärderar foderrelaterade åtgärder för att minska utsläpp och scenarier för utökad hävd av betesmarksskötsel och relaterade ekosystemtjänster inom givna metangränser. Att inkludera ekosystemtjänster i klimatberäkningar kan ha stor effekt: upp till 48 % av emissionerna allokerades till dessa tjänster i stället för till kött och mjölk, vilket kan minska risken att multifunktionalitet går förlorad vid klimatåtgärder. Utfodring av dikor med halm och gräs-klöver-ensilage minskade inte klimatpåverkan för nötkött på grund av utsläpp från markkol och avskogning, men kunde öka matproduktionen och hävd av betesmark. 3-NOP i fodret till mjölkkor minskade klimatpåverkan för mjölk med 12 % från en intensiv gård (från 0,74 till 0,65 kg CO_2e/kg mjölk) och med 9 % för mjölk från en extensiv gård (från 0,99 till 0,91 kg CO_2e/kg mjölk). Trots större reduktionspotential i det intensiva systemet bidrog det extensiva mer till andra nyttor som betesmarksrelaterade ekosystemtjänster. Scenarier som utforskade leveransen av ekosystemtjänster och bevarande av biologisk mångfald genom ökad hävd av betesmark i Sverige visade att förändringar i dagens animalieproduktion (som kastrering av tjurar och uppfödning av dem som stutar i extensiva system) kan öka arealen betesmark utan ökade metanutsläpp. Om metanutsläppen tilläts öka med 10 %, kan arealen betesmarker fördubblas. Sammanfattningsvis visar avhandlingen hur klimatpåverkan från svensk nötkötts- och mjölkproduktion kan bedömas och minskas, samtidigt som multifunktionalitet beaktas, och understryker behovet av ett systemperspektiv som integrerar utsläpp, matproduktion och ekosystemtjänster.

Keywords: nötkreatur, livscykelanalys, ekosystemtjänster, betesmarker, alternativa foder, fodertillskott, hållbara livsmedelssystem, markanvändning

Dedication

To those who strive, in big ways or small, to make the world better

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. von Greyerz K., Tidåker P., Karlsson J.O. & Röös E. (2023). A large share of climate impacts of beef and dairy can be attributed to ecosystem services other than food production. Journal of Environmental Management. 116400. doi:10.1016/j.jenvman.2022.116400

II. von Greyerz K., Ericsson N., Jardstedt M., Hessle A., Arvidsson Segerkvist K. & Röös E. (2024). Feeding straw to suckler cows spared land but did not decrease the climate impact of beef (2024). Renewable Agriculture and Food Systems. 2024;39:e36. doi:10.1017/S1742170524000255

III. von Greyerz K., Ericsson N., Ramin M., Fant P., & Röös E (submitted). Climate impact mitigation potential in milk from intensive and extensive dairy systems using 3-NOP: A life cycle perspective (under review).

IV. Karlsson J.O., von Greyerz K., Hessle A., Lindberg M., Glimskär A., Tidåker P., Bengtsson J. & Röös E. (submitted). Scenarios for long term conservation of Swedish semi-natural grasslands with limited climate impact. (under review).

The contribution of Karin von Greyerz to the papers included in this thesis was as follows:

I. Conceived the study together with the co-authors. Performed the climate impact assessment, collected data and performed the analysis. Wrote the paper with input from the co-authors.

II. Conceived the study together with the co-authors. Performed the climate impact assessment, collected data and performed the analysis. Wrote the paper with input from the co-authors.

III. Conceived the study together with the co-authors. Performed the climate impact assessment, collected data and performed the analysis. Wrote the paper with input from the co-authors.

IV. Helped plan the workshop, analyzed and compiled the results from the workshop, collected and processed data for scenarios, contributed to the manuscript on scenario descriptions, and provided input to the manuscript.

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Abbreviations

CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalents
CW	Carcass weight
DM	Dry matter
ECM	Energy-corrected milk
FPCM	Fat- and protein-corrected milk
GHG	Greenhouse gases
GWP	Global warming potential
ha	Hectare
ICBM	Introductory Carbon Balance Model
LCA	Life cycle assessment
M	Million
NDF	Nitrous oxide
N_2O	Neutral detergent fiber
t	Metric tons
3-NOP	3-nitrooxypropanol

1. Introduction

There is an urgent need to reduce greenhouse gas (GHG) emissions in response to global warming and the ongoing climate crisis. In the European Union agricultural emissions, i.e. enteric fermentation, manure management and emissions from soils, account for around 10% of total emissions. Of these, roughly two-thirds come from enteric fermentation and manure management, where cattle make the greatest contribution (European Environment Agency 2025). This highlights both a clear need and a potential for emission reductions within cattle production.

Most emissions from cattle production originate from enteric fermentation, but additional emissions arise from other parts of the production chain, such as feed production and manure management. There are several measures to mitigate these emissions, some of which are related to feed (Gerber et al. 2013). One mitigation option is to use feeds with lower climate impacts (Holtshausen et al. 2021). Another option is CH_4-reducing feed additives, which have shown promising results (Hodge et al. 2024). Such measures must be assessed from a lifecycle perspective, as they may shift emissions to other areas of the production system. There are lifecycle studies that have investigated such mitigation options within cattle systems (e.g. Alvarez-Hess et al. 2019; Dorca-Preda et al. 2024; Thomas et al. 2025), but none have been conducted within a Swedish context comparing intensive and extensive systems.

Although cattle systems have a high climate impact, they can also be multifunctional, providing several values to society. This thesis focuses on two dimensions of multifunctionality: the contribution to ecosystem services (ES) and upgrading of biomass.

Cattle can provide several ecosystem services. For example, in addition to food, grasslands used for cattle feed are associated with multiple ecosystem services, such as carbon storage and opportunities for recreation. Some grasslands are also important for biodiversity conservation, which supports ecosystem services such as pollination (Bengtsson et al. 2019; Zhao et al. 2020). In Sweden, ruminants such as cattle play an indispensable role in maintaining semi-natural pastures through grazing. These pastures, grasslands that have a long history of being neither cultivated nor fertilized except through grazing, are important to preserve, as they host many endangered species (Eriksson & Cousins 2014).

Regarding upgrading of biomass, cattle can act as “upgraders”, producing human-edible food from coarse biomass that humans can not directly digest (e.g., grass, straw, and agricultural by-products) (Karlsson 2022). This can reduce the need for cropland to grow feed crops (Mottet et al. 2017), allowing this land to be used for other purposes, such as food production for direct human consumption.

Emissions of GHG from cattle products are commonly assessed using life cycle assessment (LCA). However, results from lifecycle studies can vary widely depending on methodological choices and differences in production systems. Emissions can differ substantially from farm to farm depending on factors such as feed types, animal lifespan before slaughter, and the extent to which animals graze. When cattle systems are assessed, it is usually only food products that are considered (e.g., de Vries et al. 2015; Baldini et al. 2017; Clune et al. 2017). This often results in beef and milk produced in more intensive systems, characterized by high yields, large proportions of concentrate feeds, and limited grazing, having lower climate impacts than products from more extensive, grazing-based systems. Extensive systems typically rely on higher proportions of forage, involve longer animal lifespans, and therefore lower productivity. This subsequently results in higher emissions per unit of product (Capper 2012; Ogino et al. 2016; Bragaglio et al. 2018). Therefore, if the multifunctionality of cattle systems is not considered in the assessment, intensive systems risk being favored based on their lower climate impact. Such an approach then risks other valuable contributions being overlooked, such as the maintenance of semi-natural pastures and the biodiversity they support.

While product-level assessments, such as LCA, focus on individual products, national-level models can also be used. Through these, the national feasibility of different strategies when multiple scenarios are assessed together can be evaluated, thereby capturing system-level interactions and trade-offs.

2. Aim and structure

2.1 Aim and objectives

The general aim of this work is to enhance knowledge about how climate impacts from beef and milk production can be quantified and reduced, while considering the multifunctionality of these systems and exploring trade-offs under varying scenarios. More specific objectives are to:

1. Investigate how accounting for multifunctionality affects the assessment of the climate impact of beef and milk (Paper I).
2. Quantify the climate impacts for beef and milk using feed-related mitigation measures in Swedish cattle systems (Papers II and III).
3. Assess how multifunctionality influences the estimated mitigation potential of a CH_4-reducing feed additive in cattle production (Paper III).
4. Investigate the potential to support multifunctionality by expanding the grazing of semi-natural pastures under CH_4 emission constraints (Paper IV).

2.2 Structure of research and thesis

This thesis is based on four papers (Papers I–IV). Figure 1 illustrates the main focus of each paper (accounting for multifunctionality and climate change mitigation) and the scale of the assessments (product level and landscape level).

Paper I focuses on how the methodology to account for multifunctionality affects the climate impact assessment (Objective 1). Here, non-provisioning ecosystem services provided by cattle systems are accounted for as system outputs. This attributes climate impacts not only to food products, but also to these services, demonstrating how this affects the climate impact of cattle beef and milk at the product level.

Papers II and III address the potential for climate change mitigation at the product level. In these studies, different feed-related strategies are investigated (Objective 2). Paper II evaluates the use of straw, a by-product from wheat production, as feed, including the value of upgrading this non-edible biomass into food (beef). Paper III assesses the mitigation potential of the feed additive 3-NOP in reducing emissions from enteric fermentation. The assessment focuses on two contrasting dairy systems, one extensive and one intensive, with varying multifunctionality, demonstrating how the multifunctionality influences the mitigation potential (Objective 3).

Lastly, Paper IV extends the assessment to a landscape and system level by modelling scenarios for Swedish agriculture as a whole. In this paper, a range of strategies are evaluated to increase grazing on semi-natural pastures, and thereby non-provisioning ecosystem services, while climate mitigation is addressed by applying different limits on enteric CH_4 emission (Objective 4). The non-provisioning ecosystem services are accounted for as system outputs, quantifying the area of managed semi-natural pastures.

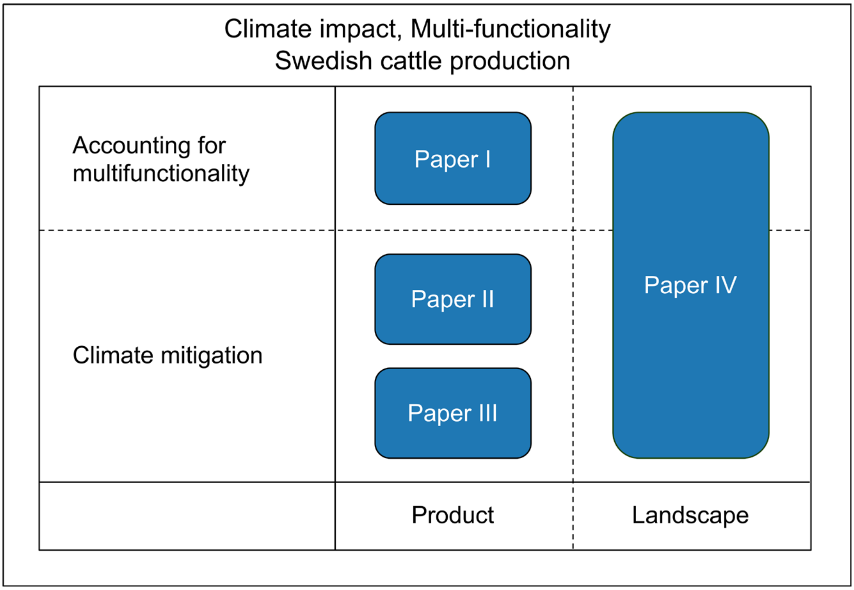

Figure 1. Thematic illustration of the thesis structure.

3. Background

3.1 Climate impact from cattle systems

Emissions of GHGs from ruminant production predominantly consist of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), which arise from processes such as enteric fermentation, manure management, feed production, and energy use. A major proportion of total GHG emissions from ruminant production originates from enteric fermentation, where CH_4 is produced during ruminal feed digestion (Gerber et al. 2013). In Sweden, enteric CH_4 emissions alone accounted for approximately 3.3 Mt of CO_2 equivalents (CO_2e) in 2022, corresponding to about 7% of the country's total GHG emissions that year, based on 100-year global warming potentials (GWP) (Swedish Environmental Protection Agency 2024).

The climate impacts of milk and beef production vary widely globally. Clune et al. (2017) reported emissions ranging from 0.54 to 7.5 kg CO_2e per kg of milk and 11 to 110 kg CO_2e per kg of bone-free beef. In the European Union, where ruminant production systems are generally more intensive, reported emissions are typically lower: approximately 17–42 kg CO_2e per kg of bone-free beef and 1–2.3 kg CO_2e per kg of milk (Lesschen et al. 2011). Nevertheless, these values are quite high compared to most other food products per kg of product and protein (Clune et al. 2017; Poore & Nemecek 2018). This is primarily due to high CH_4 emissions and relatively low feed efficiency. In Sweden, the average climate impact of beef is estimated at approximately 19 kg CO_2e per kg of carcass weight (CW) at the farm gate (Ahlgren et al. 2022). The global average is around 20 kg CO_2e per kg (based on Clune et al. 2017). On a per-kilogram protein basis, Poore & Nemecek (2018) reported global mean emissions of 50 kg CO_2e for beef from beef herds, 17 kg CO_2e for beef from dairy herds, and 3.2 kg CO_2e for milk.

The climate impact of ruminant products also varies substantially across production systems and management practices (Clune et al. 2017; Poore & Nemecek 2018). Beef and dairy systems, which are characterized by high yields or productivity and achieved by a high proportion of concentrate feed use, low slaughter age, and limited or no access to pasture, generally produce food products with a lower climate impact than low-productivity, grazing-based systems where animals live longer (Capper 2012; Kiefer et al. 2015; Ogino et al. 2016). However, reported climate impacts may also vary depending on the methodological choices that are used to assess the climate impact.

3.1.1 Strategies to decrease the climate impact

Various mitigation strategies to decrease the climate impact of beef production have been studied. For example, Cusack et al. (2021) conducted a meta-analysis of 57 comparative LCA studies, focusing on management measures to lower the environmental impacts. In that study, strategies for land-based carbon sequestration (such as adding carbon-rich amendments) showed the highest potential to reduce beef emissions with an average of 46%. Strategies to increase production efficiency (such as improved feed or supplements) showed a lower potential, with an average reduction of 8%. Strategies that combined efficiency and carbon sequestration (such as feedlots rather than conventional low-intensity grazing) reduced beef emissions by around 30%. For Western Europe, the same meta-analysis highlighted improved feed and supplements, integrated field management, and changes in fertilizer use, particularly reductions in inorganic fertilizers, as strategies to lower the climate impact (Cusack et al. 2021). In addition, several feed additives are currently being developed to reduce enteric CH_4 emissions from ruminants, e.g., 3-nitrooxypropanol (3-NOP) (Hodge et al. 2024).

One way to reduce emissions from cattle systems per kg of food product is to increase production efficiency (Cusack et al. 2021). According to Arndt et al. (2022), efficient mitigation strategies include increasing feed intake, which resulted in higher weight gain and higher milk yield, and reducing forage-to-concentrate ratio of the feed. Both strategies increased milk yield and daily weight gain, thereby reducing emission intensity. When modelling a full implementation of increased feed levels combined with a CH_4 inhibitor (e.g., 3-NOP), the same study estimated a 14% reduction in global enteric CH_4 emissions.

3.2 Multifunctionality of cattle systems

3.2.1 Ecosystem services

As ruminant systems are multifunctional to varying degrees they can provide a variety of benefits to society, including ecosystem services. The

Millennium Ecosystem Assessment (2005) defines ecosystem services as "…benefits people obtain from ecosystems". These services can be categorized into provisioning services (products, e.g., food) and non-provisioning services (non-material benefits), which include regulating services (benefits provided by regulating ecosystem processes, e.g., climate regulation), cultural services (cultural experiences, e.g., recreation), and supporting services (ecosystem functions that support other ecosystem services, e.g., soil formation). Ruminants provide the provisioning service of food, i.e., beef and milk, and can also support non-provisioning ecosystem services (Food and Agriculture Organization of the United Nations 2016). For example, grasslands used for ruminant feed provide carbon storage (regulating ecosystem services) and recreational opportunities (cultural ecosystem services). These grasslands also contribute to biodiversity conservation, which in turn supports ecosystem services such as pollination (Bengtsson et al. 2019; Zhao et al. 2020). In Sweden, ruminants play an important role in maintaining semi-natural pastures, which are only cultivated or fertilized through the animals' grazing. These pastures have significant natural, cultural, and historical values and play a key role in promoting biodiversity by providing a habitat for endangered species, which have declined over time (Eriksson & Cousins 2014). Other management practices can also provide ecosystem services. Native cattle breeds contribute to genetic values and cultural heritage. Moreover, smaller farms enhance landscape diversity and support small biotopes, providing additional ecosystem services (Karlsson et al. 2022). In regions where cash crop production is limited, such as northern Sweden, livestock are important for maintaining open landscapes, which are valued for their aesthetics, by preventing abandonment or overgrowth of agricultural land.

3.2.2 Upgrading of biomass

Another valuable aspect of ruminants is their ability to convert biomass that is unsuitable for human consumption, such as grass and straw, into nutrient-dense food for humans. This is made possible by the ruminant digestive system's capacity to extract energy and nutrients from cellulose-rich material (Karlsson 2022). When using such feed sources, cattle systems have been considered beneficial from the following perspectives (e.g. Schader et al. 2015; Mottet et al. 2017; van Hal et al. 2019a):

Contributions to food supply: Utilizing biomass that would otherwise not enter the food supply increases overall food production without requiring additional cropland. van Hal et al. (2019a) showed that limiting livestock to feed solely on 'low-opportunity-cost feeds' (food waste, food processing byproducts, and grass resources) in the European Union can produce up to 31 g of animal protein per capita and day. Cattle fed on food processing byproducts and grass resources, provided 610 g of milk and 33 g of beef per capita and day.

Reducing feed-food competition: According to Mottet et al. (2017), livestock production uses about 40% of global cropland, and producing 1 kg of ruminant meat requires, on average, 2.8 kg of feed suitable for human consumption. This contributes to the so-called feed–food competition, as cropland that could be used to grow food for direct human consumption is instead used for feed production, reducing the efficiency of the food system. Thus, reducing the use of human edible feed in ruminant production can enhance resource efficiency.

Reduced environmental impacts: Limiting livestock to non-food competing feedstuffs can reduce environmental impacts, such as GHG emissions, nutrient losses, and resource use, mainly due to fewer animals and less fertilizer demand (Schader et al. 2015; Röös et al. 2016; Karlsson & Röös 2019). It can also decrease environmental impacts from feed production (Lindberg et al. 2021; Fang et al. 2023), however the results depend on the feed type and methodological choices. For instance, Lindberg et al. (2021) showed that replacing a commercial protein feed with one based on by-products reduced land use, eutrophication, and climate impact, but increased energy use due to the need for drying. Moreover, the results of the climate impact assessment were dependent on whether emissions from deforestation were included.

3.3 Life cycle perspective

3.3.1 Life cycle methodology

According to ISO 14040:2006, LCA is a method used to evaluate the environmental impacts of a product or service across all processes throughout its entire life cycle. An LCA consists of the following 4 phases (Swedish Standards Institute 2006a):

- Definition of goal and scope
- Inventory analysis
- Impact assessment
- Interpretation

During the goal and scope phase, the aim of the LCA is established, including the study's purpose and intended use. The scope outlines the study's frame and methodological approach. During the inventory analysis, data on the system's inputs and outputs are collected and calculated. In the impact assessment phase, the results from the inventory analysis are aggregated to evaluate potential environmental impacts. During the interpretation phase, the findings from the impact assessment are analyzed and conclusions are drawn, addressing limitations and providing recommendations. Conducting an LCA is an iterative process, meaning that results from one phase can influence other phases and may lead to adjustments throughout the study (Swedish Standards Institute 2006a).

When the system has several valuable outputs, the impacts must be distributed among them. This can be achieved through various methods. According to the ISO 14044:2006, the method for this should be selected following a specific hierarchy. First, allocation should be avoided by either (i) subdividing the system into more subprocesses, or (ii) expanding the system to include the co-products, a procedure known as system expansion. If this is not possible, the impacts should be (iii) allocated according to physical relationships, meaning that the impacts are distributed proportionally to their physical relationship, such as mass or volume, or (iv) based on the economic value of the products, which is referred to as economic allocation (Swedish Standards Institute 2006b).

Another important modelling choice in LCA is the so-called 'functional unit'. The functional unit is a quantitative measure that represents the primary function of the studied system that the environmental impacts will be related to (Swedish Standards Institute 2006a).

3.3.2 Life cycle assessment of cattle production systems

LCA is a common method to evaluate the environmental impacts of food products, including those from cattle production systems. The results of such assessments can vary depending on several methodological choices. One important aspect is deciding which outputs to include. Impacts are allocated between these outputs, meaning that the choice of outputs will influence the results. According to the ISO 14044:2006, all valuable outputs from the system should be included in the allocation (Swedish Standards Institute 2006b). For cattle systems, however, it is common to only consider the food products as valuable outputs, and all associated impacts are therefore solely attributed to these products. However, cattle systems can provide additional valuable functions, such as ecosystem services (see section 3.2).

For cattle systems, economic allocation is frequently used, although system expansion and allocation based on physical relationships are also applied. For example, the International Dairy Federation (2015) recommends a biophysical allocation method from Thoma et al. (2013) for dairy systems, in which impacts are allocated to milk based on net energy for growth and milk production, and on the production of milk and body mass. The choice of allocation method influences the results, as do the specific details of the allocation (Cederberg & Stadig 2003), such as the economic values assigned to the different outputs (Guinée et al. 2004).

Moreover, various functional units are used in LCA of cattle systems. The most common is a mass- or volume-based functional unit (e.g. kg of milk or meat). Other functional units are nutrition-based (e.g. kg protein or kcal) or land-based (e.g. ha). These different functional units provide different results and can favor different types of production systems (Masset et al. 2015; Hashemi et al. 2024). When using a product-based functional unit, products from more efficient systems tend to have lower emissions per unit of product than those from extensive systems. However, if a land-use-based functional unit is used instead, systems with lower productivity can result in less impact per functional unit (Hashemi et al. 2024).

3.3.3 Climate impacts using life cycle methodology

An LCA typically considers several environmental impact categories. However, it is also common to apply life cycle methodology in studies that focus exclusively on climate impact, such as carbon footprint studies. There is also a separate ISO standard (Swedish Standards Institute 2018) for this which explains the requirements and guidelines for quantifying the carbon footprint.

Models from the IPCC guidelines for national GHG inventories are frequently used in LCA for food production to quantify on-farm emissions (e.g., Jianyi et al. 2015; Kiefer et al. 2015), although they are primarily developed for national GHG reporting. The guidelines provide models and emission factors for estimating GHG emissions from various sectors, including agriculture, forestry, and other land uses. Within these guidelines, emissions can be modelled at different levels of methodological complexity, referred to as tiers, where Tier 3 represents the most detailed approach. However, higher-tier methods require more extensive and detailed data, which can make them more demanding to apply. The use of a higher tier is often recommended when sufficiently detailed data are available or when emissions from a specific category (e.g., CH_4 from enteric fermentation or N_2O from manure management) stand for a significant share of the total emissions (IPCC 2019a).

To compare different GHGs and aggregate them into one result for climate impact, different metrics can be used. A commonly used metric for this is GWP with a 100-year time horizon (GWP100). GWP quantifies the amount of radiative forcing caused by a pulse emission of a GHG (that is, a single emission occurring at one point in time) over a specified time horizon. GWP uses CO_2 as the reference gas and expresses how much cumulative radiative forcing a given emission would cause compared to the same mass of CO_2. To express emissions in CO_2e, each gas is assigned a specific GWP factor that reflects its relative radiative efficiency and atmospheric lifetime (IPCC 2021).

3.4 Considering multifunctionality in climate impacts assessments from cattle systems

LCA studies on cattle systems generally tend to focus on the food produced, excluding other values that these systems provide, such as non-provisioning ecosystem services from the maintenance of semi-natural pastures. There are several approaches to account for ecosystem services. For example, Ahlgren et al. (2022) have used a scoring method for assessing biodiversity, adding on to life cycle assessed environmental impact categories, such as climate impacts. Thus, one approach to considering these values is to complement climate impact assessments with other methods to obtain a more comprehensive assessment (Bergez et al. 2022).

Some studies have used an alternative approach, integrating more ecosystem services directly into the climate impacts. In this approach, additional values provided by ruminant systems, such as non-provisioning ecosystem services, are treated as valuable outputs together with the food products, thereby also allocating emissions to these values. This can substantially affect the climate impacts of beef and milk products (Ripoll-Bosch et al. 2013; Kiefer et al. 2015; Bragaglio et al. 2020).

To account for feed-food competition, van Hal et al. (2019b) suggest a food-based allocation approach. Unlike physical or economic allocation, where impacts can be shared between the food products and their eventual by-products unsuitable for human consumption, food-based allocation assigns all impacts to the food products, leaving the by-products burden-free. For example, when applied to egg production where the laying hens were fed low-opportunity-cost feeds, such as bakery rest streams, this method reduced the estimated climate impact by 57% per kg of eggs.

3.5 CIBUSmod

CIBUSmod is a biophysical mass-flow model that assesses how changes in production or demand affect resource use and environmental impacts, such as GHG emissions, land use, and nutrient flows, in agri-food systems at both the national and sub-national level (Karlsson et al. 2025). Using user-defined scenarios, the model quantifies crop and livestock production and allocates it across 106 regions through an optimization procedure that restricts deviations from current production while applying agronomic constraints, such as crop rotations and regional feed availability, to generate realistic and feasible outcomes.

A baseline representing the Swedish agri-food system is included in the model based on data from 2016–2020, including e.g. statistics on consumption, trade, land use, livestock numbers, and agricultural production (Karlsson et al. 2025).

4. Methods

The studies in this thesis used different modelling approaches. To quantify the climate impacts, Papers I-III used a life cycle perspective whereas Paper IV used a biophysical mass-balance model. Paper I assessed the climate impacts of beef and milk production for ten real farms, where non-provisioning ecosystem services were treated as system outputs. Impacts were allocated to beef, milk, and these non-provisioning services using economic allocation. Paper II assessed the climate impacts of beef from a theoretical beef farm, comparing two different winter feeds for suckler cows: a mixture of grass–clover silage and straw, and grass–clover silage alone. Paper II also examined the resulting effects on food production on the farm, and the area of semi-natural pastures maintained by grazing. Paper III assessed the climate impacts of milk from two theoretical dairy farms using the CH_4-reducing feed additive 3-NOP.

Paper IV used the biophysical mass-balance model CIBUSmod to model scenarios for enhanced management of semi-natural pastures. The area of semi-natural pastures managed nationally in Sweden was quantified while limiting enteric CH_4 emissions to three different levels (−30%, 0%, and +10% from the current level). The climate impact of food and increases in national food production were also studied.

4.1 Farm descriptions

In this thesis, both real and theoretical farms were modelled. Paper I studied 10 real Swedish cattle farms representing diverse production systems. The farms varied in location, bovine density, feeding practices, and levels of beef and/or milk production. They were grouped into five categories based on different production systems:

- Suckler systems: Four farms that breed calves from suckler cows, both of which are kept on the farms, where one farm also fattens purchased suckler calves.
- Dairy calf systems: Two farms that fatten calves purchased from neighboring dairy farms.
- Mixed suckler & dairy calf system: One farm that breeds suckler calves and fattens calves from dairy farms.

- Dairy systems: Two farms that specialize in milk production, with beef from culled cows and surplus calves sold for fattening.
- Dairy & dairy calves system: One farm that produces milk and beef from culled cows and fattens surplus dairy calves.

In Paper II, one theoretical farm was used instead. This was based on the beef farm in the Götaland forest district described in Ahlgren et al. (2022), that fattens calves from suckler cows for beef, keeping both on the farm.

Paper III focused on milk production and also used theoretical farms. To assess how multifunctionality influences the effect of 3-NOP, two contrasting dairy systems were studied: an intensive farm oriented toward high milk yield and an extensive farm managed to provide values beyond food production:

- Intensive farm: High milk yield and located in a region with large herds, high productivity, and limited semi-natural pastures.
- Extensive farm: using a native cattle breed and located in a region with natural constraints and higher proportions of pastures, and typically small herds.

The key contrasts in management strategies and associated multifunctionality outcomes of these farms are summarized in Table 1.

Table 1. Key differences in management and multifunctionality between the studied intensive and extensive farms in Paper III

Management practice	Farm comparison	Related function/value
Grazing period on semi-natural pastures for dairy cows and replacement heifers	3 months for extensive farm; none for intensive	Management of semi-natural pastures
Forage fraction in feed	Higher in extensive than intensive	Coarse biomass upgraded to food
Keeping of native breed	All cattle on extensive farm; none in intensive	Conservation of native breed
Milk production	Higher per cow and year on intensive farm	Food production
Use of land in less favorable areas	Extensive farm located in less favorable area	Landscape openness
Grass clover mixtures in feed	Included on both farms	Cultivation of grass-clover leys
Farm size	Extensive smaller than intensive	

Paper IV did not adopt a farm perspective; instead, the Swedish agricultural system was studied as a whole. This means that the model used in this study not only included cattle systems but also other livestock (both ruminant and monogastric), as well as both the production of feed and food crops.

4.2 Considering non-provisioning ecosystem services in the climate impact assessments

In Paper I, non-provisioning ecosystem services were included in the assessment by considering them as additional valuable outputs, thereby allocating climate impacts not only to food products but also to these services. Emissions were allocated among the different outputs using economic allocation, whereby total emissions were proportionally distributed according to the economic value of each output. For food products, producer prices were used as the basis for valuation. For non-provisioning ecosystem services, which do not generate a direct market income, compensatory payments through agri-environmental schemes were used as a proxy for their economic value, representing society's economic valuation of certain ecosystem services, as previously done by e.g. Ripoll-Bosch et al. (2013). The relationship between these payments, specific ecosystem services, and livestock production is not always straightforward, as payments vary in how directly they are associated with ecosystem services and livestock. For example, payments for maintenance of semi-natural pastures are more directly connected to the livestock, as grazing animals are necessary for their management, whereas payments for feed production are less directly connected, since these can also be granted for cultivation of non-feed crops. This complicates decisions regarding which payments to include in the assessment. To examine how different assumptions regarding the coupling of ecosystem services-related payments to livestock production affect outcomes, payments were divided into three groups, with additional payments included depending on their connection to the livestock production (Figure 2).

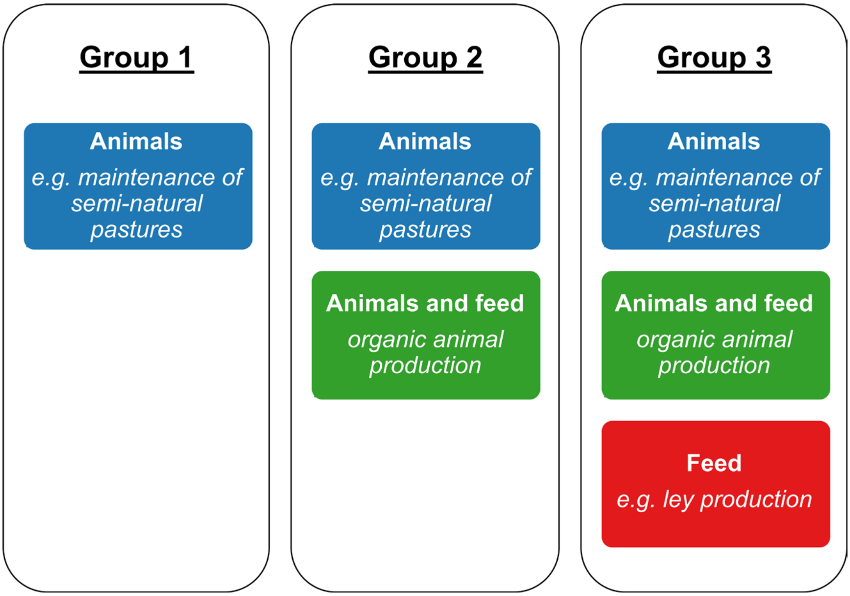

Figure 2. Payment groupings used for allocation to non-provisioning ecosystem services.

4.3 Climate impact mitigation measures

Two feed-related options for decreasing the climate impacts were studied: partially replacing a commonly used winter feed for suckler cows, grass-clover mixtures, with cereal straw (Paper II) and using the CH_4-reducing feed additive 3-NOP in dairy production (Paper III).

4.3.1 Mixture of grass-clover silage and straw

Suckler cows in Sweden typically graze during summer and are housed indoors during winter, where they are predominantly fed grass-clover silage (Ahlgren et al. 2022). However, allowing unrestricted access to early-cut forage, which is a common management practice, can result in overfeeding. Substituting part of this feed with more fiber-rich forage, such as cereal straw, could reduce winter feed intake and thus potentially lower both the environmental impact and feed costs (Jardstedt 2019).

Therefore, Paper II compared two winter feeding strategies for suckler cows: a combination of grass-clover silage and straw versus grass-clover silage only. The inclusion of straw reduced silage intake, which decreased the area of cropland needed for feed production. Two alternative uses for this spared cropland were considered: i) to produce wheat for human consumption, and ii) to convert it into additional pasture. Due to the lower nutrient intake of winter feed, suckler cows that were fed straw during winter were assumed to increase their grass intake during grazing to compensate. The increased grazing intake raised pasture demand. When the spared cropland was converted into pasture, this demand was met. When the spared cropland was used to produce wheat, new pasture had to be established on what was assumed to be previous forest land. These variations in winter diet and cropland use resulted in three scenarios:

- Reference: Cows fed only grass-clover silage during winter.
- Straw-food: Cows fed a mixture of straw and grass-clover silage during winter. The spared cropland was used to grow wheat for human consumption, and additional pasture was established from forest land to meet grazing needs.
- Straw-pasture: Cows fed a mixture of straw and grass-clover silage during winter. The spared cropland was converted into pasture to cover increased grazing demand.

4.3.2 3-Nitrooxypropanol

In the rumen, microbes ferment dietary carbohydrates, generating hydrogen and CO_2, which are normally converted into CH_4 by microbes in the rumen via an enzyme (Buccioni et al. 2015). 3-NOP, a feed additive developed to reduce enteric CH_4, inhibits this enzyme, thereby directly reducing CH_4 production in the rumen (Duin et al. 2016). 3-NOP is commercially available as Bovaer®, a powder administered through feed (dsm-firmenich n.d.), which limits its practical use during grazing. Bovaer® is considered safe for dairy cows with a maximum recommended dose of 100 mg per kg dry matter (DM) (EFSA Panel on Additives and Products or Substances used in Animal Feed (FEEDAP) et al. 2021). Studies have shown that 3-NOP can lower CH_4 emissions from enteric fermentation by about 30% on average, with e.g., reductions ranging from 15 to 65% per cow per day, depending on the dose and chemical composition (fat, neutral detergent fiber (NDF) of the diet (Kebreab et al. 2023).

In Paper III, 3-NOP supplementation was assumed to be given to dairy cows exclusively during the lactation period, and thus not administered during the dry period. Within the intensive production system, cows were assumed to receive 3-NOP for 10 months per year, corresponding to a 2-month dry period. In the extensive system, the supplementation period was set to 9 months, due to both the 2-month dry period and an additional month during lactation when cows grazed on semi-natural pastures without supplementary feeding. In both production systems, the dose was set at 60 mg 3-NOP per kg DM per cow, since higher doses can have a negative effect on milk production and DM intake (Martins et al. 2025). The percentage reductions in enteric emissions were calculated using the formula for CH_4 production (g/d) from Kebreab et al. (2023).

4.4 Scenarios for increasing the maintenance of semi-natural pastures

Paper IV explored several scenarios aimed at expanding grazing on semi-natural pastures at a national scale in Sweden under different methane caps (-30%, 0% and +10%). The scenarios were compared with a baseline representing current Swedish agriculture (around the year 2020). Preliminary scenarios with potential to increase grazing on semi-natural pastures were identified, and the scenarios were later revised based on feedback from a workshop involving stakeholders and researchers. The final scenarios and their deviations from the baseline are presented in Table 2. The population of pigs and poultry was kept constant across all scenarios and the baseline, as were the ratios between beef and milk production and between beef and lamb production, except in the WinLamb scenario. In this scenario, lamb production was allowed to increase relative to beef production to offset higher lamb imports and align better with actual meat consumption. The amount of cropland in each region was also kept constant.

Table 2. Description of the scenarios investigated to expand semi-natural pastures and their deviation from the baseline (Paper IV)

Scenario	Baseline description	Scenario description
MaxCur	Current livestock systems graze semi-natural pastures as today.	Current livestock systems graze semi-natural pastures at their maximum potential.
Steers	Grazing is limited for most male cattle for slaughter, as they are typically raised as intact bulls and kept indoors.	All male cattle for slaughter are castrated and managed as low-intensity steers, largely grazing semi-natural pastures.
CulCows	Dairy cows are culled after their final lactation (unless removed earlier due to injury or disease).	Dairy cows scheduled for culling pre-grazing are dried off and retained to graze semi-natural pastures for another season.
DryCows	Dairy cows typically have a two-month dry period, during which they graze semi-natural pastures until one month before calving.	Dairy cows have a four-month dry period, with three months grazing on semi-natural pastures.
WinLamb	Around 20% of Swedish lambs are raised as winter lambs, grazing in summer after spring birth and housed until late winter or early spring.	All lambs are raised as winter lambs since this system has the highest potential for grazing semi-natural pastures.
RecHorses	Horses obtain only a limited portion of their grazing intake from semi-natural pastures.	Breeding horses, ponies, and cold-blooded horses are assumed to graze semi-natural pastures extensively.
All		All scenarios above combined.

0.51 million hectares of semi-natural pastures were considered as currently managed, and 1.2 million hectares as could potentially be managed. Thus, the managed area could be increased by around 0.7 million hectares.

4.5 Climate impact assessment

4.5.1 System boundaries and functional unit

Papers I-III

In Papers I-III, life cycle methodology was used to assess the climate impact from the systems under study (see section 3.3.1). A cradle-to-gate perspective was used, meaning that processes and emissions up until the animals left the farm for slaughter were included in the assessment. The processes used in the different studies are presented in Figure 3.

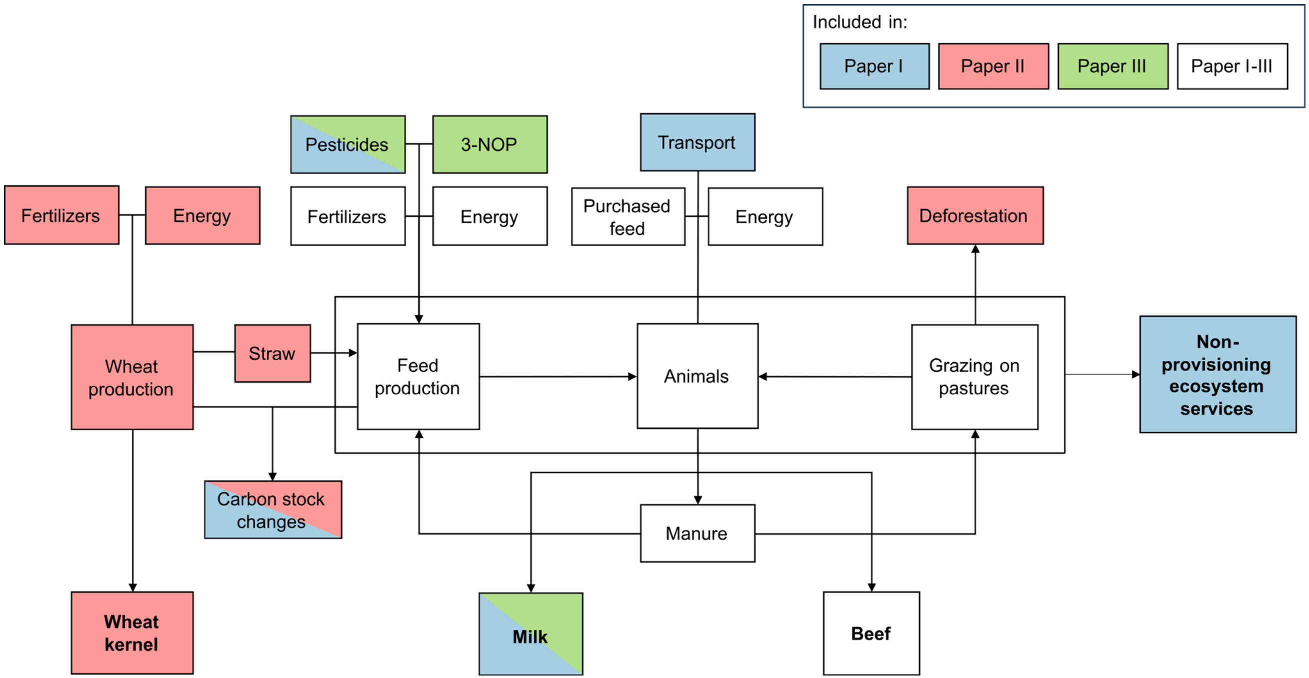

Figure 3. System boundaries used for the assessments in Papers I-III.

Emissions from the production and maintenance of capital goods were not included. Emissions from inputs used in smaller volumes, such as medicines, were also excluded. The functional unit was set to 1 kg CW for beef and sold calves and 1 kg fat- and protein-corrected milk (FPCM) for milk in Paper I, 1 kg CW for beef in Paper II, and 1 kg energy-corrected milk (ECM) for milk in Paper III.

Paper IV

In Paper IV, the CIBUSmod model was used (see section 3.5). The model included the following processes: enteric fermentation, manure management, energy use on farms (e.g., in stables and field machinery), soil processes related to crop production (e.g., from application of manure and crop residues), production of inputs (e.g., fertilizers and energy), and waste management (Karlsson et al. 2025). The emissions were presented per kg protein and per m^2 of the managed semi-natural pastures.

4.5.2 Emission modelling

This thesis considers only climate impact, including emissions of CO_2, CH_4, and N_2O. The different GHGs were converted into CO_2e using GWP100, as stated in Table 3.

Table 3. Global warming potential (GWP100) factors used in the climate impact assessments

GWP100 factors	**Paper I (IPCC 2013)**	**Papers II-IV (IPCC 2021)**
Carbon dioxide	1	1
Biogenic methane		27.0
Fossil methane		29.8
All methane	34	
Nitrous oxide	298	273

In Papers I–III, GHG emissions were primarily modelled using IPCC (2019a) methodologies, following the Swedish national inventory report. In Paper I, CH_4 emissions from cattle were modelled using a Tier 2 approach (IPCC 2019b), whereas Papers II and III applied a Tier 3 approach (Bertilsson 2016), based on DM intake and content of fatty acids in feed for cows, and gross energy content and protein content in feed for other cattle.

In Papers I-III, CH_4 and N_2O from manure management, N_2O from manure added to pastures from grazing animals, and N_2O from fertilizers, manure, and crop residues on cropland were modelled using a Tier 2 approach from IPCC (2019b; c). N_2O from manure is based on nitrogen excretion from animals (calculated from nitrogen content in feed and nitrogen retention in animals) and manure management system or grazing (IPCC 2019b). Emissions of N_2O from fertilizers, manure, and crop residues on cropland are based on their nitrogen content (IPCC 2019c). Emissions from production inputs, such as energy and fertilizers, were modelled with emission factors from different literature sources (e.g. Flysjö et al. 2008; Gode et al. 2011; Wernet et al. 2016).

Soil carbon stock changes on cropland were included in Papers I and II, modelled using the Introductory Carbon Balance Model (ICBM; Andrén et al. 2004) as a Tier 3 approach. This model estimates annual changes in topsoil soil carbon from initial stocks and yearly carbon inputs. Soil carbon is described using first-order kinetics and is divided into a young and old pool. All new carbon enters the young pool, where the majority is mineralized into CO_2, while a fraction is transferred to the old pool. The young pool is further divided by input type (aboveground residues, belowground biomass, and manure). Pool dynamics are set by decomposition rates, input-specific humification factors, as well as decomposer activity factors, which consider soil temperature, moisture, and cultivation intensity (Andrén et al. 2008), chosen by the user.

Subsequently, CO_2 emissions from changes in soil carbon were quantified by comparing scenarios with a reference. In Paper I, average soil organic matter data from the same region as each farm (Paulsson et al. 2015) were used as the reference. In Paper II, the carbon stock changes for the straw scenarios were calculated as the difference between the reference scenario and straw scenarios. Soil carbon stock changes were modelled over a 30-year period, with a yearly average used in the climate impact assessment. Additionally, Paper II included emissions from potential deforestation to convert forests into semi-natural pastures. This was estimated based on the average timber stocks for the Götaland region (Swedish University of Agricultural Sciences 2022). The removed tree biomass was assumed to contain 50% carbon, which was released as CO_2. These emissions were distributed over 100 years.

In Paper IV, the emissions were modelled with CIBUSmod. This model also included emissions of CO_2, CH_4, and N_2O, where the emissions were calculated using models from IPCC (2019a), as well as from the European Environment Agency (2023) and Bertilsson (2016).

4.5.3 Allocation of emissions

For beef and milk in Papers I and III, economic allocation was used, based on producer prices to allocate emissions between beef and milk. Economic allocation was also used to account for additional values in the assessments (non-provisioning ecosystem services in Paper I; see section 4.2).

In Paper II, only the emissions from wheat were allocated, as only beef (and no milk) was produced. Here, all wheat emissions were allocated to the grain, leaving the straw burden free. This followed the approach suggested by van Hal et al. (2019b) who stated that such an allocation more successively accounts for feed-food competition by assigning no environmental burden to biomass streams that are unfit or undesirable for human consumption. In line with this reasoning, straw, which is considered a by-product without a direct food function for humans, was treated as a burden-free input when upgraded to beef production systems.

In Paper IV, no allocation of emissions was performed, since emissions were calculated at a national level.

4.6 Food and nutrient production

In Paper II, the output of food (beef products and wheat) and their energy, fat, and protein content was assessed as an additional indicator to evaluate how the straw-based suckler cow diet affected the system's contribution to food production. Annual meat production was estimated from the number of animals slaughtered and their slaughter weights (Ahlgren et al. 2022). By-products such as offal and blood were included, with fractions used for current human consumption based on Strid et al. (2022). Wheat production was based on yield and cultivated area, with fractions of wheat kernels currently used for human consumption based on Amcoff et al. (2012) and Tillgren (2021). Energy, protein, and fat contents were obtained from the Swedish food database (National Food Agency 2023). Nutrients from animal products (beef, offal, blood) and wheat were assessed separately due to differences in quality and digestibility (Joye 2019; Bajželj et al. 2021).

In Paper IV, the increased grazing also led to less cropping of winter feed. The cropping of cereals was used as a proxy for the potential to increase food production, calculated by multiplying the cropland area spared from the reduced need for winter feed with regional cereal yields.

4.7 Inventory sources

In Paper I, primary data from actual farms were used, complemented with secondary data, while Papers II–IV relied on only secondary data, such as Swedish statistics (e.g., Växa (2023); Swedish Board of Agriculture (n.d.)) and reports (e.g., Ahlgren et al. (2022)). Region-specific data were applied when available (e.g., crop yields), otherwise, national data were used (e.g., energy use in stables). Feed is a key factor in assessing the climate impact of cattle systems, since both its production and nutritional composition influence GHG emissions as well as the mitigation strategies considered in this thesis. In Paper I, annual on-farm feed use was reported by farmers, and feed intake per animal was estimated using IPCC (2019b) formulas. In Papers II and III, feed intake for the straw scenarios, for dairy cows in both intensive and extensive systems, and for recruitment heifers in the extensive system was calculated by experts using established models (Spörndly 2003; NorFor n.d.). The feed intake for the remaining animals and scenarios was based on literature data.

5. Results and discussion

5.1 Inclusion of non-provisioning ecosystem services in the assessment (Paper I)

When non-provisioning ecosystem services were not included, the climate impact of beef varied between 16 and 39 kg CO_2e per kg CW, and for milk it varied between 0.65 and 1.2 kg CO_2e per kg FPCM (Figure 4). Beef from suckler farms (Suckler A-D and Suckler and dairy calves A) had the highest impact because all emissions from suckler cows were allocated to beef. For beef originating from dairy production, a large proportion of emissions were instead allocated to milk production. Animals on suckler farms also had higher slaughter ages, leading to greater CH_4 production. The farm producing both milk and fattened calves (Dairy and dairy calves A) showed higher climate impacts for both beef and milk than the other dairy farms. This was due to extensive on-farm calf fattening, lower milk yield, and a larger share of emissions allocated to beef from the higher income that was obtained by fattening calves on the farm.

Climate impact for beef, kg CO_2e per kg CW	Excluding non-provisioning ecosystem services	Including non-provisioning ecosystem services		
		Group 1	Group 2	Group 3
Suckler A	39	35 (-11%)	28 (-29%)	27 (-31%)
Suckler B	36	31 (-13%)	31 (-27%)	26 (-28%)
Suckler C	35	22 (-36%)	18 (-47%)	18 (-48%)
Suckler D	33	29 (-13%)	24 (-26%)	23 (-30%)
Suckler and dairy calves A	31	25 (-18%)	22 (-28%)	21 (-31%)
Dairy calves A	22	22 (0%)	22 (0%)	22 (-3%)
Dairy calves B	22	22 (0%)	22 (0%)	22 (-1%)
Dairy and dairy calves A	21	18 (-13%)	16 (-23%)	15 (-31%)
Dairy A	16	16 (-3%)	15 (-8%)	15 (-11%)
Dairy B	16	15 (-4%)	14 (-10%)	13 (-18%)

Larger change

Smaller change

Climate impact for milk, kg CO_2e per kg FPCM	Excluding non-provisioning ecosystem services	Including non-provisioning ecosystem services		
		Group 1	Group 2	Group 3
Dairy and dairy calves A	1.2	1.1 (-13%)	0.94 (-23%)	0.85 (-31%)
Dairy A	0.65	0.63 (-3%)	0.60 (-8%)	0.58 (-11%)
Dairy B	0.96	0.93 (-4%)	0.87 (-10%)	0.79 (-18%)

Larger change

Smaller change

Figure 4. Climate impacts of beef per kg carcass weight (CW) (upper) and milk per kg fat- and protein-corrected milk (FPCM) (lower) using the different payment groups for allocation (Paper I).

Including non-provisioning ecosystem services in the assessment substantially influenced the results, with up to 48% of emissions attributed to non-provisioning ecosystem services rather than to beef and/or milk (Figure 4). The magnitude of this reallocation varied depending on the type of payment considered. Payments most directly associated with animals (i.e., for maintenance of semi-natural pastures, keeping endangered domestic animal breeds, maintenance of semi-natural pastures with special values in areas of natural constraints, and organic animal husbandry) had the strongest influence, reallocating up to 36% of emissions to non-provisioning ecosystem services. Payments targeting organic livestock farming also had an effect (added in Group 2), shifting up to 18% of emissions. In contrast, payments related to feed production had only a minor effect, with a reallocation of up to 8%. This suggests that payments more directly connected with ruminant production have a greater role in reallocating emissions to non-provisioning ecosystem services, based on how society values these services through the payment system. However, this also reflects policy decisions influenced by multiple priorities beyond non-provisioning ecosystem services and thus may not directly correspond to the actual provision of these services.

Regarding comparisons of the farms, differences in the climate impact of beef and milk remained large when only Group 1 payments were included, whereas including more payments reduced the difference in the food products' climate impact between farms, especially Group 2. Thus, it is important to consider which payments are included.

The allocation method had a substantial impact on the results for several farms. However, the different farm types were affected to varying degrees. The impact of beef from suckler farms, which initially had the highest emission intensities when non-provisioning ecosystem services were not considered, was the most affected. In contrast, beef from farms rearing surplus dairy calves was the least affected. On the suckler farms, extensive management with low inputs and high pasture use led to slower growth and later slaughter. This reduced food production per hectare while enhancing other ecosystem services, making these farms more affected by the allocation method. Beef from dairy farms was less affected than that from suckler farms, even though their income from non-provisioning ecosystem services could be similar. Because milk generates a higher income per hectare than beef, allocation factors were less influenced by the income from the non-provisioning ecosystem services, leading to a smaller shift for dairy farms.

5.2 Mitigation measures

5.2.1 Straw as winter feed for suckler cows (Paper II)

The climate impact from feed production was 3.7, 3.4, and 3.2 per kg CW for the reference, straw-food, and straw-pasture scenarios, respectively (Figure 5). Including straw in the winter feed for suckler cows thereby reduced emissions from feed production compared with using silage only per kg CW, with reductions of 7% and 15%, respectively. From a lifecycle perspective, the climate impacts were 25, 27, and 25 kg CO_2e per kg CW for the reference, straw-food, and straw-pasture scenarios, respectively (Figure 5). When accounting for changes in soil carbon stocks and potential deforestation-related emissions (caused by the need for more pasture during summer), beef from the straw-food scenario had even higher overall emissions than the other scenarios. This shows that using byproducts as feed does not always lower the climate impact. Grass-clover leys contribute more to soil carbon than cereals; their replacement with wheat and straw removal in the straw-food scenario led to soil carbon losses and net CO_2 emissions from soils. The conversion to low-yield pasture in the straw-pasture scenario also led to soil carbon losses and CO_2 emissions. Additional emissions resulted from potential deforestation to establish new pastures. This could be avoided by creating pasture through other means, such as converting low-yielding cropland, as was the case in the straw-pasture scenario. However, converting cropland that could produce food into pasture may intensify competition between food and feed production.

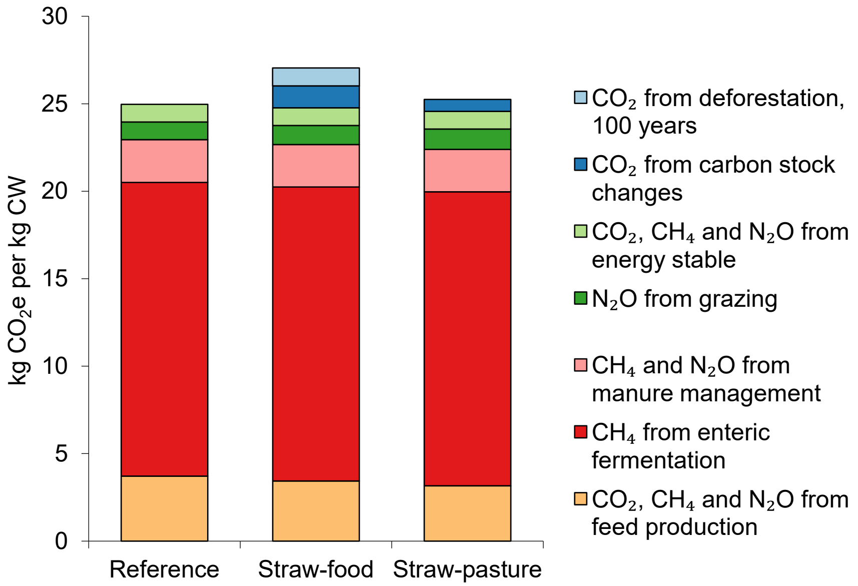

Figure 5. Climate impact of beef per kg carcass weight (CW) from the different scenarios in Paper II, separated into carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) (Paper II).

When straw was used as feed, feed intake was lower during winter and less cultivated feed was therefore required, releasing cropland for alternative uses. Using this land to grow wheat substantially increased the amount of wheat and macronutrients (up to 80%, Table 4). Moreover, the addition of straw to suckler cow diets during winter increased grazing intake during summer, enabling the same number of cows to manage a larger pasture area. When no deforestation was needed to expand pasture, additional pastures could be managed without increasing GHG emissions.

Table 4. Beef, wheat kernels, and macronutrients produced in the different scenarios. Lower values represent current utilization rates, whereas higher values assume full utilization (Paper II)

	Scenario		
	Reference	Straw-food	Straw-pasture
Beef, t carcass weight	12	12	12
Wheat whole grain, t	27	48	27
Nutrients from meat, offal, and blood			
Energy, GJ	83-87	83-87	83-87
Protein, t	2.7-2.9	2.7-2.9	2.7-2.9
Fat, t	0.96-1.0	0.96-1.0	0.96-1.0
Nutrients from wheat			
Energy, GJ	220-380	380-680	220-380
Protein, t	1.3-2.7	2.3-4.8	1.3-2.7
Fat, t	0.30-0.68	0.54-1.2	0.30-0.68

5.2.2 3-NOP (Paper III)

Without 3-NOP, the climate impact was 0.74 and 0.99 kg CO_2e per kg ECM for the intensive and extensive systems, respectively. With 3-NOP supplementation, the climate impact was reduced to 0.65 and 0.91 kg CO_2e per kg ECM, respectively (Table 5). From a life cycle perspective, this corresponded to a reduction of 12% and 9% per kg ECM, respectively, where the higher percentage reduction per kg ECM also reflects the initially lower emissions per kg. Emissions associated with the production of 3-NOP accounted for only 0.24% and 0.18% of total emissions, indicating that the additive substantially mitigates more emissions than it generates.

Milk from the intensive system had a lower climate impact than milk in Moberg et al. (2019), which showed a Swedish average impact of 1.1 kg CO_2e per kg ECM. This was, e.g., due to higher milk yields, less use of fertilizers, and differences in manure management.

Table 5. Methane emissions and total climate impact per kg ECM in intensive and extensive farms, with and without 3-NOP (Paper III)

	Intensive		**Extensive**	
	Without 3-NOP	With 3-NOP	Without 3-NOP	With 3-NOP
Methane emissions per kg ECM	0.44	0.35	0.58	0.49
Emissions from production of 3-NOP per kg ECM		0.013		0.0021
Total climate impact per kg ECM	0.74	0.65	0.99	0.91

Management practices and, in turn, multifunctionality, influenced the efficiency of the mitigation potential of 3-NOP. The key differences in management, multifunctionality, and their effect on 3-NOP efficacy are summarized in Table 6.

Table 6. Differences in management and multifunctionality and how they influenced the effect of 3-NOP between the intensive and extensive farm (Paper III)

Function/value	**Management practice**	**Farm comparison**	**Differences affecting 3-NOP effect**	**Impact on 3-NOP efficacy**
Coarse biomass upgraded to food	Forage fraction in feed	Higher in extensive than intensive	Higher NDF content in feed used on extensive farm	Lower methane reduction during supplementation period (23% vs. 28%)
Management of semi-natural pastures	Grazing period on semi-natural pastures	3 months for extensive; none for intensive	3-NOP administrated for one month less in extensive	Lower methane reduction during lactation period (21% vs. 28)

The higher proportion of coarse biomass, and consequently NDF, in the diet of dairy cows on the extensive farm led to a somewhat smaller reduction compared to the intensive system per day during supplementation (23 vs. 28%). Moreover, the use of semi-natural pastures further limited the effect, as dairy cows on the extensive farm could not be supplemented with 3-NOP while grazing semi-natural pastures during lactation, whereas cows on the intensive farm received the additive throughout the entire lactation period. This resulted in a longer supplementation period for the intensive farm (ten months compared with nine in the extensive system). Consequently, the mitigation potential of 3-NOP on enteric fermentation emissions during the lactation period was greater in the intensive system (−28%) than in the extensive system (−21%). From a life cycle perspective, i.e., including other emission sources that were included in the assessment, the emission reduction was similar (12% vs. 9%), which suggests that multifunctionality did not substantially affect the mitigation potential of 3-NOP for milk. The overall climate impact, however, was lower in the intensive farm both before and after 3-NOP supplementation, primarily due to its higher efficiency in producing more milk per cow.

5.3 Semi-natural pastures expansion under limited methane emissions (Paper IV)

When including all scenarios (see Table 2), an expansion of managed semi-natural pastures was possible under each CH_4 cap (Figure 6). Allowing CH_4 emissions to increase by 10% enabled a more than twofold increase in managed semi-natural pastures (up to 1.1 Mha). In contrast, a 30% reduction in CH_4 emissions limited the managed area to 0.7 Mha, which was still a 38% increase from current levels. These results show it is possible to increase the area of semi-natural pastures while maintaining or even reducing the number of livestock.

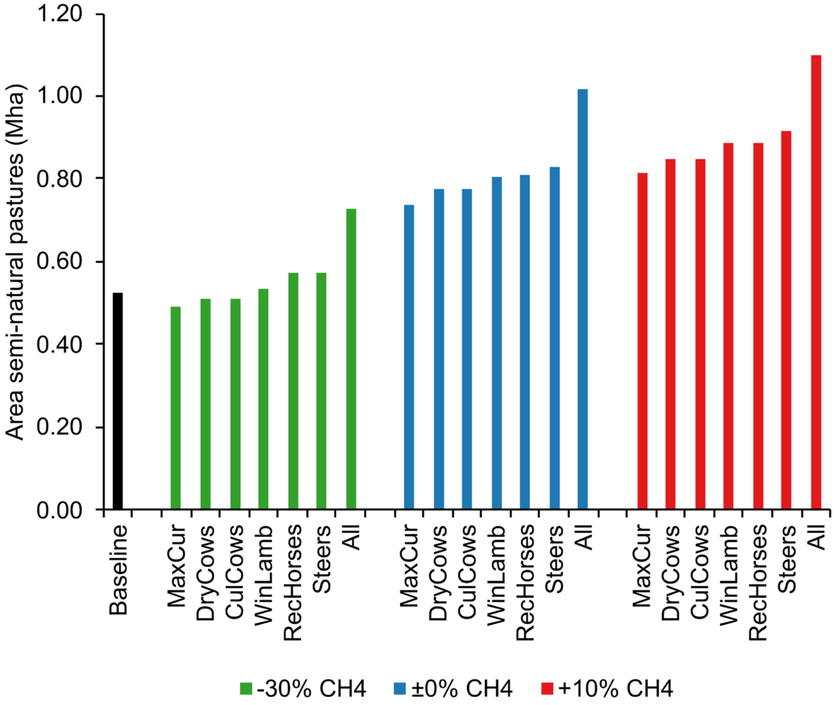

Figure 6. Potential semi-natural pasture grazing from the different scenarios and methane (CH_4) limits in Paper IV (Paper IV).

Among the individual scenarios, castrating all male cattle for slaughter and managing them as low-intensity steers primarily grazing semi-natural pastures had the greatest potential to increase grazing on semi-natural pastures, with up to 0.4 Mha more than the baseline. In contrast, scenarios involving changes in dairy production showed the lowest potential. The combined effect of all scenarios was the largest.

Although a larger area of semi-natural pastures could be managed while CH_4 emissions were reduced by 30% overall, this led to higher emissions per kg of protein and per hectare of pasture (Figure 7). This was predominantly driven by lower cattle and sheep production, while the number of horses increased; it was assumed that horses did not contribute to food production but generated emissions.

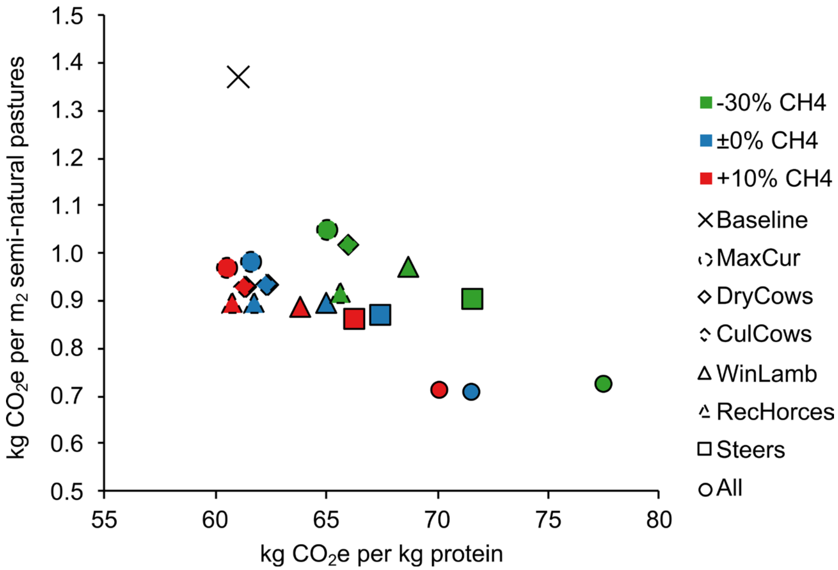

Figure 7. Greenhouse gas emissions per kg protein and per semi-natural pasture area for the different scenarios and methane (CH_4) limits in Paper IV (Paper IV).

The interventions for increased grazing resulted in a reduced demand for cropland when grazing was expanded. If the released cropland was then used for cereal production, the cereal output would increase by 45%, 16%, and 8% under the −30%, 0%, and +10% CH_4 emission caps when implementing the all scenario. This was due to increased grazing of semi-natural pastures rather than arable land, and, under a reduced CH_4 cap, a lower number of animals (20% fewer livestock units) further reinforced it.

6. General discussion

6.1 Climate impact mitigation

Various mitigation strategies can be applied to reduce GHG emissions from cattle production. In Paper III, the use of 3-NOP showed potential to reduce milk emissions by around 10%. If all dairy cows in Sweden were given 3-NOP under the conditions described in Paper III, annual emissions of enteric CH_4 could be reduced by 0.26-0.34 Mt CO_2e per year, corresponding to approximately 4-5% of Swedish agricultural emissions. Further emission reductions would be possible by increasing the dose of 3-NOP. However, this may reduce DM intake and milk yield (Maigaard et al. 2025). Additional mitigation is possible by giving the additive to heifers. At present, 3-NOP is only approved for dairy cattle (EFSA Panel on Additives and Products or Substances used in Animal Feed (FEEDAP) et al. 2021), but research has shown that it can also reduce emissions among beef cattle which could additionally lower national CH_4 emissions (Orzuna-Orzuna et al., 2024).

Another way to reduce emissions from cattle products is to increase efficiency, i.e., increase beef and milk yields (Arndt et al. 2022). However, beef and milk from systems that provide additional values beyond food production had higher emissions than those from more intensive systems, which has also been reported in previous studies (Capper 2012; Ogino et al. 2016; Bragaglio et al. 2018). This therefore risks reducing the other values, e.g., non-provisioning ecosystem services. For example, increasing efficiency with finishing beef in feedlots generally has lower climate impacts than beef produced from cattle on grazed pastures (Cusack et al. 2021), but this change would reduce the potential non-provisioning ecosystem services provided by pastures.

An alternative strategy is to apply emission reduction measures that do not negatively affect multifunctionality. For example, the use of 3-NOP reduced CH_4 emissions from milk by around 10% without affecting the other values provided by the system. If approved for beef cattle, this could be combined with interventions to further increase grazing while allowing more semi-natural pastures to be grazed. If an efficient method to give the additive to animals during grazing cannot be found, the CH_4-mitigating potential for suckler cows and their calves will be lower due to longer grazing periods. However, it can still have an important effect. Similarly, CH_4-reducing feed additives could be used with straw as feed to e.g. provide the benefit of increased grazing of semi-natural pastures which would compensate for the eventual increase in emissions when substituting some ley for straw in the diet. In addition to 3-NOP, other mitigation measures that were not investigated in this thesis (e.g., see section 3.1.1) could potentially be combined with the measures investigated here to further reduce emissions (Arndt et al. 2022).

Even if the climate impact per kg product increases, total emissions from the agricultural sector can still be reduced. One way to achieve this is by reducing the number of ruminants. For instance, in Paper IV, the highest climate impact per kg protein was reported for the CH_4 cap where emissions were lowered by 30%, which reduced the number of ruminants due to their enteric CH_4 emissions. Another example is outlined in the study by Resare Sahlin et al. (2022), in which an intensive farm transitioned to a more extensive production system, resulting in a higher climate impact per kg of beef, while total farm-level emissions decreased due to a reduced number of animals. This transition also reduced resource use, such as feed inputs, and had positive effects on other sustainability dimensions. Reducing animal numbers while allowing production systems to deliver additional values beyond food production can therefore lower agricultural-sector emissions while maintaining, or even enhancing, other values.

6.2 Implementation challenges

Implementing both straw feeding and 3-NOP faces several challenges. At present, 3-NOP can only be provided by mixing it into feed, preventing supplementation when animals are grazing without supplementary feed. Alternative delivery methods, such as through drinking water (Rivelli et al. 2025), could potentially enhance mitigation in extensive systems.

For straw feeding, daily alternation between silage and straw bales, a common feeding strategy, often leads to overconsumption on days when silage is fed. This can cause digestive problems. Mixing the two in a feed wagon ensures a higher and steadier intake of straw. However, this method requires equipment that small farms in particular may be unable to afford, thereby rendering it challenging for implementation. This is particularly important in Sweden, where most suckler herds consist of less than 50 cows (Swedish Board of Agriculture 2022).

Including 3-NOP in cattle diets will also raise production costs, therefore its adoption may depend on economic incentives such as consumer price premiums or subsidies (Swedish Climate Policy Council 2025). Consistent with the findings from Papers II and III, the workshop in Paper IV also identified economic conditions as a major constraint, such as increased costs for inputs and logistics. Thus, to make these changes possible, economic incentives are necessary.

At the time of writing, in Denmark, larger dairy farms are required to implement CH_4-reducing feeding strategies, such as 3-NOP. Farms subject to this regulation may apply for financial compensation (Styrelsen for Grøn Arealomlægning og Vandmiljø 2025). Another option is to raise beef and milk prices to offset the increased cost. However, this approach depends on consumers' willingness to pay.

Consumer and farmer perceptions can also present a challenge to the implementation of mitigation strategies. Regarding 3-NOP, there are negative perceptions among some consumer groups and farmers concerning e.g., the fear of chemical residues in the milk and manipulation of a natural process, which has resulted in boycott movements (Manning et al. 2025). 3-NOP is also, at the time of writing, prohibited in organic production in European Union (European Commission implementing regulation (EU) 2021/1165). Considering that 18% of Swedish dairy cows are in organic production, this hinders a substantial share from receiving the additive.

Agri-environmental payments are already available for climate mitigation and biodiversity conservation through the European Union's common agricultural policy (European Commission n.d.), including support for restoring and maintaining semi-natural pastures. However, the workshop on scenarios for increased grazing on semi-natural pastures performed for Paper IV indicated that payments for semi-natural pastures do not fully cover the associated increase in production costs. Consequently, payments would need to be higher to be effective according to the participants.

6.3 Interventions' effect on multifunctionality

Changes in production, such as using mitigation measures, can affect system multifunctionality. The use of 3-NOP did not affect the multifunctionality of the system, only the climate impact. However, the changes investigated in Papers II and IV, which involved increasing the upgrading of biomass, i.e., partly using straw as winter feed for suckler cows and implementing scenarios for enhanced management of semi-natural pastures, affected the area of semi-natural pastures managed and food production.

Both papers demonstrated that the managed semi-natural pastures can be expanded with livestock numbers being kept the same or even decreased, which is beneficial from a climate mitigation perspective. This aligns with Larsson et al. (2020), who suggested that the number of animals is not the limiting factor for the upkeep of semi-natural pastures in Sweden, but rather economic factors. Limiting livestock to non-food competing feedstuffs (such as biomass from semi-natural pastures) can reduce environmental impacts since non-food competing resources are limited, thereby restraining animal numbers (Schader et al. 2015). These management changes also showed a possibility of increasing the production of biomass that could be used as food, consistent with van Hal et al. (2019a). This showed that utilizing biomass that would otherwise not contribute to the food supply increases food production without increasing the use of cropland, because cropland used for feed production can be used to grow food crops.

However, in Paper IV, nearly all scenarios resulted in lower protein production compared to the baseline, since most of the scenarios resulted in lower production efficiency when CH_4 levels were not allowed to increase. Depending on the scenario, milk and ruminant meat output declined by 32–43%. In contrast, when CH_4 emissions were permitted to increase by 10%, both protein production and the use of semi-natural pastures could increase, although not in all scenarios. Similarly, in Paper II, the scenario in which food production (Straw-food) was increased by using the spared land to grow more wheat also increased emissions from kg CW of beef. In this case, it was due to the deforestation emissions from expanding pasture to meet the grazing demand and emissions from carbon stock changes. However, the amount of beef produced did not change.

6.4 Methodological choices and trade-offs

When focusing solely on product-based climate impacts, there is a risk of favoring more intensive production systems, which may have lower emissions per kg of product but provide fewer non-provisioning ecosystem services. In the scenarios for expanding the grazing on semi-natural pastures (Paper IV), the lowest CH_4 cap (-30%) led to higher emissions per kg of protein and per hectare of pasture compared to the other CH_4 caps. Additionally, the scenarios with a higher climate impact per kg of protein had a lower climate impact per hectare of semi-natural pasture. This demonstrates a trade-off between the climate impact of food products and non-provisioning ecosystem services from maintenance of semi-natural pastures.

Moreover, this trade-off demonstrates that the choice of functional unit can influence the interpretation of the results which can consequently affect policy decisions. Previous studies have reported similar trade-offs where, for example, conventional systems were favored with a product-based functional unit while organic systems were favored when the unit is area-based. Thus, one way to highlight the trade-off can be to use both product and area-based units (Hashemi et al. 2024). According to the ISO 14040:2006, the functional unit should reflect the function of the system (Swedish Standards Institute 2006a). Since one function of cattle systems is to produce food, the product-based unit is suitable. However, the non-provisioning ecosystem services, e.g., from maintenance of semi-natural pastures, are also a function of the system. Therefore, the area of semi-natural pastures, as used in Paper IV, can serve as an important complement.

In Paper I, the trade-offs between climate impact and non-provisioning ecosystem services were addressed by including the non-provisioning ecosystem services directly within the product climate impact. This provides a simple way of capturing multiple perspectives with a single metric, instead of using several indicators or assessments. Such an approach can support various purposes, including consumer communication and decision-making.

For example, using non-ecosystem services-adjusted climate impacts to incentivize emission reductions among organic farmers could unintentionally encourage intensification, potentially compromising animal welfare and biodiversity (Röös et al. 2018). Using this approach could help mitigate that risk. However, when the purpose of the study requires trade-offs to be more visible, an approach that shows both impacts and values would be preferable, e.g., scoring for biodiversity in addition to climate impacts (Ahlgren et al. 2022), which this approach does not.

In this thesis, this approach was applied only in Paper I, where around 30% of the climate impacts of beef from most suckler farms were reallocated to non-provisioning ecosystem services. Thus, the results for the beef from the suckler farm used in Paper II might have been similarly affected if this method had been applied. Since the use of straw led to more semi-natural pastures being grazed, the climate impacts of beef from the straw scenarios might also have been more affected than in the reference scenario, with a larger share of emissions reallocated. In Paper I, this approach also shifted parts of milk and beef impacts to non-provisioning ecosystem services for the dairy farms, where the most extensive farm was affected the most. Consequently, using that approach could have reduced the differences in climate impacts from milk observed in Paper III.

Another allocation issue concerns deforestation from restoring semi-natural pastures. In Paper II, all the deforestation emissions were allocated to beef production, leading to a higher climate impact in the scenario with deforestation. However, it could be argued that these emissions should instead be allocated to the restoration of semi-natural pastures as a separate process. Such an approach would have influenced the comparison between scenarios, making their climate impacts more similar. In Paper IV, deforestation emissions were excluded. If the expansion of semi-natural pastures were to cause deforestation and those emissions were allocated to the food produced, emissions per kg protein would increase. The increase would then be the greatest for higher CH_4 caps, as these led to a larger expansion of semi-natural pasture area.

To reduce the risk of decreasing multifunctionality, a key question is how to handle trade-offs in climate impact assessments. However, it is important to note that changes in assessment methodologies (such as in Paper I) should not be conflated with a reduction measure. Absolute emissions remain unchanged and must still be addressed when working toward climate targets, e.g., by using reduction measures.

6.5 Uncertainties and methodological choices

6.5.1 Inventory data

In Paper I, primary data from real farms were used, complemented with secondary data. Data collected directly from farms represent real farming conditions but may not be completely reliable since farmers do not always keep precise records of all data needed for LCA. For example, only the total amount of feed used per year was reported, and for some farms, the reported forage amounts appeared unrealistic, e.g., high amounts of forage use compared to the requirement. These values were therefore recalculated using farm information together with feed models and estimates of feed losses found in literature (Hessle et al. 2017; IPCC 2019b). Thus, the results may not fully reflect the actual conditions on the farm. For example, if the recalculated values underestimate feed losses, the overall estimates of feed use and related emissions would also be underestimated. This is an important aspect in climate impact assessment of cattle products, as feed production accounts for a larger share of the impact.

Papers II-IV only used secondary data, such as statistical averages and normative yields from official databases. Farming conditions vary considerably between years and farms, which introduces variability in parameters. For example, crop yields fluctuate due to factors such as climate variability (Ray et al. 2015). Moreover, variations in feed nutritional quality can influence estimates of enteric CH_4 emissions as well as the mitigation effects of 3-NOP supplementation. Consequently, the results are subject to considerable variation, which should be taken into account.

6.5.2 Methane modelling

Several models exist for estimating enteric CH_4 emissions, which can lead to varying results. The models used in Papers II-IV (Nielsen et al. 2013) are recommended for Swedish systems (Bertilsson 2016) and applied in the Swedish National Inventory Report (Swedish EPA, 2021a). For example, estimated emissions for the suckler cows in Paper II would be about 20% higher if an alternative model from Nielsen et al. (2013) had been used instead. However, since the same models were applied across scenarios within each study, model uncertainties are shared across scenarios. While this may influence absolute emissions, the effect on the comparisons is expected to be marginal.

Enteric CH_4 reductions from 3-NOP per animal per day were estimated using an empirical model (see Kebreab et al. 2023 for uncertainties). Given that both systems in this study relied on the same model, this level of uncertainty for these calculations should have a limited influence on their relative comparison.

6.5.3 Soil carbon stock modelling

In Papers I and II, changes in soil carbon stocks were modelled, which involves considerable uncertainty. Carbon inputs, i.e., crop residues and manure, are a key factor in these models and represent one of the largest sources of uncertainty (Menichetti et al. 2024). Different models are available to estimate these inputs, resulting in varying outcomes, particularly for ley cropping (Keel et al. 2017). In addition, ley yields are especially uncertain because it is not common for farmers to weigh the harvested ley (as is the case with cereals) (Cederberg et al. 2018). This further contributes to the uncertainty, particularly regarding the farms in this thesis, which were fed a high proportion of ley.

6.5.4 Deforestation

In Paper II, deforestation emissions were included and estimated using average timber stocks in the study region, assuming a carbon content of 50% in the biomass emitted as CO_2. The deforestation emissions, however, depend on forest characteristics such as land coverage, which vary widely (Swedish University of Agricultural Sciences 2022). The selected allocation period also impacts the results; using a 100-year period, as applied here, yields lower annual emissions than shorter allocation periods (e.g., 20 years). Emissions from soil carbon changes due to deforestation were excluded, as they were assumed to be small within this context and highly uncertain (Bárcena et al. 2014), though such effects can vary with soil type, climate, and management. These uncertainties and methodological choices can affect the emissions. However, only one scenario led to deforestation (Straw-food), which remains the highest-emitting scenario even when deforestation was not taken into account.

6.5.5 Economic allocation

In Papers I and III, economic allocation was used to divide the climate impact between the different outputs (beef, milk, and non-provisional ecosystem services in Paper I and beef and milk in Paper III). The ISO 14044:2006 standard for LCA notes that economic allocation should be used when other methods are not suitable (Swedish Standards Institute 2006a). Alternative allocation methods were considered unsuitable for handling the multifunctionality of meat, milk, and ecosystem services, as in Paper I. There is no common physical denominator (e.g., mass) for these outputs, nor is there an appropriate system that would allow the use of system expansion.

Ultimately, economic allocation is associated with uncertainties (Ardente & Cellura 2012). For example, market prices used to set the product's economic value can fluctuate with time (Marvuglia et al. 2010), as can subsidies. This also makes it difficult to validate and compare the results of different studies (Marvuglia et al. 2010). However, it has been claimed that economic allocation more effectively captures societal drivers responsible for the emissions (Ardente & Cellura 2012). For example, in life cycle assessments of rapeseed oil, using mass allocation assigns a larger share of the environmental impact to rapeseed cake, a byproduct of production. In contrast, economic allocation assigns most of the impact to the oil (Fridrihsone et al. 2020), which is the main product and the primary driver of production.

6.5.6 Aggregating greenhouse gas emissions

This thesis applies GWP to compare GHG emissions, but alternative metrics and models exist. Global temperature potential, for example, quantifies the temperature effect of emissions at a specific future time (Persson et al. 2015). GWP* is an approach that estimates how changes in GHG emissions affect global temperature, by comparing current CH_4 emissions to a previous emissions level and translating the change in rate of emissions into an equivalent amount of warming. For short-lived gases, such as CH_4, this means that constant levels of emissions will result in little additional warming (Lynch et al. 2020). Several studies recommend using GWP to assess the climate impact of foods (e.g., Landquist et al. 2019; Ahlgren et al. 2022; Ran et al. 2024), while GWP* is generally considered unsuitable for product-level assessments and LCA because rather than capturing the impact of one additional unit, it reflects the effect of emissions relative to a prior level (Röös 2019).

GWP values for several timeframes are available (IPCC 2021). This thesis uses GWP100, but shorter (20-year) or longer (500-year) timeframes can produce very different outcomes. This is especially relevant for short-lived gases such as CH_4, which is important for ruminant systems. For example, Röös (2019) found that beef's climate impact per kg CW was roughly twice as high using GWP20 versus GWP100. Röös (2019) also suggests that a 100-year timeframe is appropriate for comparing food products, as it captures both CH_4's strong short-term effect and CO_2's long-term persistence, and this approach is also commonly used in carbon footprint studies.

7. Conclusions

The overall aim of this thesis was to increase knowledge on how climate impacts from beef and milk production can be quantified and reduced, while considering the multifunctionality of cattle systems and exploring trade-offs under varying scenarios. The following conclusions have been drawn from this work.

Including non-provisioning ecosystem services alongside food provisioning services when estimating the climate impact of ruminant products can have a large effect on the emissions intensity of the products. Up to approximately half of the climate impact was shifted from food products to other ecosystem services. This method can therefore help reduce the risk of losing multifunctionality in climate mitigation efforts.

Using a combination of cereal straw and grass-clover silage as winter feed for suckler cows could reduce the climate impact from feed production but does not necessarily translate into a reduced climate impact of beef. The reduction achieved from reducing the climate impact from feed production was cancelled by emissions from soil carbon stock changes (as less carbon was added to soils when the straw was removed to be used as feed and less ley was cultivated) and deforestation from the potential need to create new pastures on forested land.

Using a combination of cereal straw and grass-clover silage can lead to increased food production and increased maintenance of semi-natural pastures. As a silage-straw diet during winter results in increased grazing during summer, there is a potential to increase the grazed area. This can benefit biodiversity if biologically rich semi-natural pastures are maintained or restored. Reduced silage intake spared cropland, which could be used for other purposes, including food crop production, potentially leading to indirect climate benefits by reducing the need for food production elsewhere.

From a life cycle perspective, greenhouse gas emissions per kg of energy-corrected milk can be reduced for intensive and extensive dairy systems using the feed additive 3-NOP. The climate impact was reduced by 12% and 9% per kg of energy-corrected milk for the intensive and extensive system, respectively, where the reductions exceeded the emissions associated with the production of the additive.

The multi-functionality can influence the 3-NOP mitigation potential per cow during lactation, whereas its effect on the mitigation per kg

energy-corrected milk appears to be less pronounced. The effect of 3-NOP per cow and day was smaller on extensive farms due to a higher fraction of neutral detergent fiber in the diet, resulting from a greater proportion of coarse biomass (28% vs. 23%) and the use of semi-natural pastures. Cows grazing these pastures during lactation did not receive 3-NOP, which further limited methane mitigation during the supplementation period (23% vs. 21%). When expressed per kilogram of energy-corrected milk, the corresponding reductions were 12% and 9%, respectively.

Through changes in livestock management, the area of grazed semi-natural pastures in Sweden can be expanded without increasing methane emissions, thereby supporting biodiversity and other ecosystem services. Allowing methane emissions to increase by 10% from today's levels did, however, expand the grazed area the most, with up to 1.1 Mha being managed in total, as it permitted the highest number of grazing animals.

Increasing maintenance of semi-natural pastures through changes in livestock management can result in increased climate impact per kilogram of protein due to the lower productivity of the interventions. The climate impact per kilogram of protein was also higher when the climate impact per hectare of semi-natural pasture was lower, which illustrates a trade-off between food-system climate impact and maintaining semi-natural pastures.

Among the individual scenarios studied to expand grazing on semi-natural pastures, castrating all male cattle for slaughter shows the greatest potential. This scenario enabled up to 0.4 Mha more to be managed compared to current levels. In contrast, scenarios targeting changes in dairy production had the smallest effect.

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Popular science summary

Agriculture accounts for roughly 10% of the European Union's total greenhouse gas emissions, with cattle contributing the largest share. This highlights the importance of reducing these emissions. At the same time, these animals not only provide food in the form of beef and milk, but also other benefits. For example, grazing animals can support biodiversity and ecosystem services by maintaining species-rich semi-natural pastures. If these benefits are ignored when decreasing emissions, the benefits risk being reduced.

This thesis aims to increase knowledge about how the climate impacts of beef and milk production can be quantified and reduced while also taking these other benefits into account. It also explores trade-offs that can arise between reducing the climate impact and maintaining the other benefits. The thesis examines how ecosystem services can be included in climate impact assessments, how different feed-related changes can reduce emissions, and how grazing on semi-natural pastures and their associated benefits can be expanded while keeping methane emissions limited.

The results show that almost half of the emissions can be attributed to ecosystem services rather than to meat and milk. Therefore, including these benefits in assessments reduces the risk of them becoming lost when reducing climate impacts.

One of the feed-related changes studied was providing cows with a mixture of straw and grass-clover silage during winter. This reduced emissions from the feed itself but did not lower the overall climate impact of the beef, as it could instead lead to emissions from soil and deforestation. However, it can provide other benefits, such as increased food production and more grazing on semi-natural pastures.

The second feed-related change examined was the methane-reducing feed additive 3-NOP. Its effect on milk was assessed for two farms: one with high milk production and one with lower milk production, the latter of which provided more additional benefits, including grazing on semi-natural pastures. The climate impact of milk decreased by 12% on the high-producing farm and by 9% on the lower-producing farm. Although the climate impact for milk was lower on the high-producing farm, both when using and not using 3-NOP, the lower-producing farm contributed more to other benefits, such as biodiversity and ecosystem services.

Furthermore, the results showed that changes in current Swedish livestock production (for example, castrating bulls and raising them as slower-growing grazing steers) can increase the area of grazed semi-natural pastures without increasing methane emissions. If methane emissions are allowed to rise by 10% compared with current levels the area of grazed species-rich pastures could more than double.

In summary, this thesis shows how the climate impact of Swedish beef and milk production can be assessed and reduced while considering benefits beyond meat and milk production. The results underscore the importance of a holistic perspective that integrates greenhouse gas emissions, food production, and ecosystem services within assessments.

Populärvetenskaplig sammanfattning

Jordbruket står för ungefär 10 % av den Europeiska unionens totala utsläpp av växthusgaser, där nötkreatur står för den största andelen. Det visar att det är viktigt att minska dessa utsläpp. Samtidigt bidrar dessa djur inte bara med mat i form av kött och mjölk, utan även med andra nyttor. Till exempel kan dessa betande djur stödja biologisk mångfald och bidra till ekosystemtjänster genom att sköta artrika betesmarker. Om dessa nyttor ignoreras vid utsläppsminskningar riskerar de att minska.

Den här avhandlingen syftar till att öka kunskapen om hur klimatpåverkan från nötkötts- och mjölkproduktion kan beräknas och minskas samtidigt som de andra nyttorna beaktas. Den undersöker också avvägningar som kan uppstå mellan att minska klimatpåverkan och dessa andra nyttor. Avhandlingen undersöker hur ekosystemtjänster kan tas med i klimatpåverkansberäkningar och hur olika foderrelaterade åtgärder kan minska utsläpp. Den undersöker också hur bete på betesmarker, och därigenom dess nyttor, kan utökas samtidigt som metanutsläppen hålls begränsade.

Resultaten visar att nästan hälften av utsläppen kan tilldelas till ekosystemtjänster istället för kött och mjölk. Genom att inkludera dessa nyttor i beräkningen minskar risken att dessa går förlorade vid åtgärder för att minska klimatpåverkan.

En av de foderrelaterade åtgärderna som undersöktes för att minska utsläppen var att ge dikor en blandning av halm och gräs-klöverensilage under vintern. Detta minskade utsläppen från fodret men sänkte inte den totala klimatpåverkan för köttet eftersom detta utfodringsalternativ istället kan leda till utsläpp av markkol och avskogning. Det kunde däremot leda till andra nyttor, som möjlighet till utökad matproduktion och mer bete på betesmarker.

Den andra foderrelaterade åtgärden som studerades var det metanreducerande fodertillskottet 3-NOP. Effekten undersöktes för mjölk från två gårdar: en med hög mjölkproduktion och en med lägre, men som istället gav fler andra nyttor genom att till exempel beta artrika betesmarker. Klimatpåverkan minskade med 12 % för mjölken från den mer högproduktiva gården och med 9 % för den mer lågproduktiva gården. Klimatpåverkan var lägre både då 3-NOP användes och inte användes för mjölken från den mer högproduktiva gården. Den mer lågproduktiva gården bidrog dock till mer andra nyttor- utöver matproduktion, såsom biologisk mångfald och ekosystemtjänster

De scenarier som undersöktes för att utöka arealen betesmark som sköts i Sverige visar att förändringar i dagens djurproduktion (till exempel att kastrera tjurar och föda upp dem som långsamväxande betande stutar) kan öka arealen betade marker utan att metanutsläppen ökar. Om metanutsläppen tillåts öka med 10 % jämfört med dagens nivåer, kan arealen betade naturbetesmarker mer än fördubblas.

Sammanfattningsvis visar avhandlingen hur klimatpåverkan från svensk kött- och mjölkproduktion kan beräknas och reduceras samtidigt som multifunktionalitet beaktas. Resultaten visar på vikten av ett helhetsperspektiv, där växthusgasutsläpp, matproduktion och ekosystemtjänster integreras.

Acknowledgements

First of all, I want to warmly thank my supervisors, Elin Röös and Niclas Ericsson, for supporting me and providing me with their knowledge with me over the years. I especially appreciate your availability and all the time you have prioritized for me. Also, thanks also to my former supervisor, Dalia Abdelfattah, for her valuable support.

I would further like to thank all co-authors for their valuable help and insights, as well as everyone else who contributed knowledge and assistance to the papers and thesis.

Thanks to my colleagues at the Department of Energy and Technology for lunches, fikas, and enjoyable discussions. Especially thanks to those who have taken the time to review my work, answer questions, and prepare challenging questions for my seminars, helping me learn. I would especially like to acknowledge my PhD colleagues over the years for all the support, laughter, and good times both inside and outside the office. To those who supported me a bit extra, you know who you are, I am especially grateful.

I would also like to thank Mistra Food Futures for funding my PhD, and to its network for young researchers for interesting seminars and fun conferences.

Finally, to my family and friends, thank you for your love and support through both good times and challenging times. I am always grateful for you and so happy to have you in my life.

This PhD project was mainly funded by the Mistra Food Futures research program [Grant ID: DIA 2018/24]. Additional funding sources for the papers are listed in each paper.

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Journal of Environmental Management 325 (2023) 116400

Contents lists available at ScienceDirect

Journal of Environmental Management

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

Research article

A large share of climate impacts of beef and dairy can be attributed to ecosystem services other than food production

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ARTICLE INFO

Keywords:
Life cycle assessment
Multi-functionality
Climate impact
Economic allocation
Cattle

ABSTRACT

Domesticated ruminants supply nutrient-dense foods but at a large environmental cost. However, many ruminant production systems are multi-functional, providing ecosystem services (ES) other than direct provision of food. When quantifying the climate impact of ruminant products using life cycle assessment (LCA), provisioning ES (i.e. beef and milk) are generally considered the only valuable outputs and other ES provided are ignored, which risks overlooking positive contributions associated with ruminant production. Non-provisioning ES can be included in LCA by economic allocation, using compensatory payments (through agri-environmental schemes) as a proxy for the economic value of ES. For example, farmers can receive payments for maintenance of pastures, which supports e.g. pollination. However, the association between different payment schemes, the ES provided, and livestock production is not always straightforward and it can be difficult to determine which payment schemes to include in the allocation. This study examined how accounting for ES in quantification of climate impact for beef and milk production on Swedish farms was affected by different ways of coupling ES to livestock production through payment schemes. Quantification was done using LCA, attributing the climate impact to beef, milk, and other ES by economic allocation. This resulted in <1–48% and 11–31% of climate impacts being allocated to other ES, instead of beef and milk, respectively, affecting suckler farms most. The results were influenced by which payment schemes, representing different ES, that were included; when only payments directly related to livestock rearing were included, the difference in the climate impact was still large between farm types, while the difference decreased considerably when all environmental schemes were included. While emissions do not disappear, ES-corrected climate impact can potentially be useful as part of consumer communication or in decision-making, reducing the risk of overlooking ES provided by ruminant production in a simpler way than using separate indicators.

Author contributions

Karin von Greyerz: Conceptualization, Methodology, Formal analysis, Writing – original draft, Visualization. **Pernilla Tidåker**: Conceptualization, Methodology, Writing – review & editing. **Johan O. Karlsson**: Conceptualization, Methodology, Writing – review & editing. **Elin Röös**: Conceptualization, Methodology, Writing – review & editing, Supervision, Project administration, Funding acquisition.

1. Introduction

Food production has profound effects on ecosystems and causes major negative environmental impacts, including contributing to climate change. The agriculture sector produces an estimated 11% of anthropogenic greenhouse gas (GHG) emissions in the European Union (EU), making it the second most emissions-intense sector (European Environment Agency, 2021). Livestock production causes 81% of agricultural GHG emissions and uses approximately 65% of total agricultural land in the EU (Leip et al., 2015). Ruminants are among the livestock sector's greatest contributors to global warming, generating emissions from enteric fermentation, feed production, manure management, energy use in barns, and deforestation (Gerber et al., 2013).

However, ruminants provide nutrient-dense foods, and ruminant systems are multi-functional to varying degrees, generating other values to society (Food and Agriculture Organization of the United Nations, 2016a; 2016b). A key concept for describing such contributions is *ecosystem services* (ES), defined by the Millennium Ecosystem Assessment (2005) as *"benefits people obtain from ecosystems"* and divided into

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https://doi.org/10.1016/j.jenvman.2022.116400
Received 14 February 2022; Received in revised form 16 September 2022; Accepted 26 September 2022
Available online 18 October 2022

provisioning (i.e. production of food and other materials), regulating, cultural, and supporting ES. Many ES are now in rapid decline, requiring urgent and concerted actions to reverse this trend (IPBES, 2019). Ruminants can contribute to ES in several ways. Grasslands used to provide feed for ruminants are associated with *e.g.*, carbon storage, recreational values, and biodiversity contributing to e.g., pollination (Bengtsson et al., 2019; Zhao et al., 2020). In Sweden, ruminants are indispensable in maintenance of semi-natural pastures that are not cultivated or fertilized, other than directly by grazing animals, as these endangered ecosystems host many red-listed species (Eriksson and Cousins, 2014). However, ruminant production systems vary from intensive systems with high yields obtained using high proportions of concentrates for feed, low slaughter age, and specialized indoor production, to more extensive systems based on grazing with higher slaughter age and lower milk yields (Capper, 2012; Kiefer et al., 2015; Vagnoni et al., 2015; Ogino et al., 2016). Hence, the extent to which different livestock systems contribute to ES provisioning varies widely.

The environmental impact of ruminant production is commonly assessed using life cycle assessment (LCA) (de Vries et al., 2015; Baldini et al., 2017; Clune et al., 2017), which evaluates environmental impacts of products and services from all processes throughout the entire life cycle (SIS, 2006). LCA of milk and beef (in this study meaning beef from all cattle, including from culled dairy cows) considering the impact category of climate change (hereafter called climate impact) show results in the range 0.54–7.5 kg carbon dioxide equivalents (CO_2e) per kg milk and 11–110 kg CO_2e per kg bone-free beef globally (Clune et al., 2017). In the EU, where ruminant production is commonly more intensive, climate impacts are in the lower range; 17–42 kg CO_2e per kg bone-free meat and 1–2.3 kg CO_2e per kg milk (Lesschen et al., 2011). Still, these impacts are considerably higher than for most other comparable food products, due to high methane (CH_4) emissions and low feed efficiency. However, LCA for ruminant systems commonly include only beef and milk as valuable outputs, and not other ES potentially provided (de Vries et al., 2015; Baldini et al., 2017; Clune et al., 2017). This risks overlooking positive contributions to ES, other than direct food provisioning, when making decisions about future livestock systems. When non-provisioning ES are ignored in LCA, meat produced in intensive systems generally has lower climate impacts than meat from extensive systems (Ogino et al., 2016; Bragaglio et al., 2018). Some LCA studies have attempted to consider the multi-functionality in animal production systems using economic allocation, where the environmental impact is distributed proportionally to outputs (foods and other ES) based on their economic value (Ripoll-Bosch et al., 2013; Kiefer et al., 2015; Bragaglio et al., 2020). However, while food products can be valued using market prices, attributing value to other ES is less straightforward. To assign a monetary value to ES, previous studies have used payments through agri-environmental schemes, representing the value society assigns to certain ES. Ripoll-Bosch et al. (2013) used this method for considering cultural ES in three representative lamb production systems and found 0–46% lower climate impact for meat when ES were considered, shifting emissions from meat to other ES provided. Kiefer et al. (2015) used the same allocation method when assessing the climate impact of German milk production, considering preservation and upkeep of cultivated landscapes and preservation of endangered breeds. Payments for *e.g.* organic farming and management of biodiverse grassland and the price of milk was used to allocate emissions between the meat, milk and other ES. This economic allocation led to 1–29% of emissions being allocated to non-provisioning ES. Bragaglio et al. (2020) calculated the climate impact of beef considering biodiversity in terms of keeping local breeds and grazing on natural grasslands, conservation of landscapes, and socio-economic viability of rural areas on 25 farms in Italy divided into four clusters based on production system. That study found that accounting for ES with economic allocation shifted 0–43% of emissions from beef to other ES, i.e. milk for dairy farms and non-provisioning services e.g. services related to biodiversity conservation and cultural services.

A multitude of environmental payments schemes are available for farmers that are more or less directly associated with the delivery of ES, including support to organic farming and farming in areas of natural constraints (ANC) (farming in areas where agricultural production is more challenging due to unfavorable natural conditions). There are also a range of payment schemes more directly connected to livestock, including support for biodiversity conservation in semi-natural pastures or preservation of local livestock breeds. It can therefore be difficult to decide which payment scheme/s to include when using these as a base for economic allocation in LCA to account for non-provisioning ES, especially as payments are sometimes only vaguely reflecting the ES provided (Simoncini et al., 2019), which can give variable results.

Therefore, the aim of this study was to examine how the climate impact of beef and milk from Swedish farms representing different production systems was affected by different ways of coupling non-provisioning ES to livestock production through payment schemes. Hence, this study adds to the current literature on using economic allocation to include ES as an output in LCA by considering varying ways of including payment schemes. Quantification of the climate impact was performed using LCA for 10 Swedish cattle farms with different management practices to represent a breath in production systems while capturing the specificity provided by studying real farms. Risks and opportunities with using ES-corrected climate impact values for beef and milk in different applications were also discussed.

2. Material and method

2.1. Case study farms

Cattle production in Sweden varies from intensive dairy production with intensive breeding of dairy calves to extensive suckler production. Animal welfare regulations require outdoor grazing for all cattle except bulls, in grazing periods lasting up to 270 days, but housing periods are often long because of the harsh climate and intensive rearing. Grazing is based on leys and semi-natural pastures. Silage, cereals, and concentrates are commonly used as additional feedstuffs, with feed use differing between farms. The main feedstuff is silage harvested from grass-clover leys grown on cropland, often in rotation with other crops.

This study assessed 10 Swedish cattle farms with different production systems: Two specialist dairy farms selling surplus calves to other farms, seven pure beef-producing farms (with suckler herds and/or bought in calves), and one farm producing milk and also fattening calves for beef. The farms were all part of the Swedish case study of the Uniseco project (https://uniseco-project.eu/) and selected purposively to represent varying cattle production systems throughout Sweden. Hence, the farms represented the five production systems described below. The farms differed in terms of e.g. geographical location, feed, amounts of beef and/or milk produced, and bovine density, which are summarized in Table 1.

1. *Suckler systems*

Four farms breed calves from a herd of suckler cows, and one of these also fattens bought-in suckler calves. Most feed is produced on-farm and consists of forages and some cereals. These farms will be referred to as Suckler A, B, C and D.

2. *Dairy calf systems*

Two farms fatten dairy calves bought from neighboring dairy farms. The feed consists of forage and cereals grown on-farm and bought-in concentrates. These farms will be referred to as Dairy calves A and B.

3. *Suckler and dairy calf system*

One farm breeds suckler calves and fattens calves from other dairy

Table 1
Characteristics of the 10 Swedish farms assessed.

	Suckler A	Suckler B	Suckler C	Suckler D	Suckler and dairy calves A	Dairy calves A	Dairy calves B	Dairy and dairy calves A	Dairy A	Dairy B
Area of natural constraints	No	Yes	Yes	No	Yes	No	Yes	Yes	Yes	Yes
Organic/conventional farming	Organic	Organic	Organic	Organic	Organic	Conventional	Conventional	Organic	Organic	Organic
Farm area										
Cropland (ha)	27	27	59	51	110	48	520	150	204	80
Pastures with general values[a] (ha)	6	25	6	16	4			7	40	12
Pasture with specific values[a] (ha)	3.5		14	22	34				52	19
Bovine production										
Breed	Beef	Beef	Native[b]	Beef	Beef and dairy	Dairy	Dairy	Native[b]	Dairy	Dairy
Bovine density[c] (AU/ha)	0.65	0.45	0.21	0.72	0.42	0.63	0.84	0.36	0.60	0.78
Milk production[d] (t FPCM/y)								110	1400	550
Beef production[e] (t CW/y)	3.2	3.8	57	12	14	7.6	6000	6.3	20	6.2
Feed										
Forage (% of diet in DM)[f]	46	26	30	51	43	46	34	64	55	36
Cereals (% of diet in DM)[f]			<1		<1				11	22
Concentrates (% of diet in DM)[f]						<1	66	6	10	13
Other feed (% of diet in DM)[f]		<1				5			5	
Grazing on cropland (% of diet in DM)[f]						48				18
Grazing of semi-natural pastures (% of diet in DM)[f]	54	74	70	49	56			30	20	11
Bought-in feed (% of diet in DM)[f]				4		6	66	6	13	14
Grazing period (days)	200	270	Heifers: 180 Others: 270	180	Suckler cows: 270 Others: 180	180		125	Heifers: 180 Dairy cows: 135	Calves: 150 Heifers: 180 Dairy cows: 270
Manure management system										
Deep bedding with no mixing (%)	5	100	90	100	40	100	17	100	9	5
Solid storage (%)			10		20		38			
Liquid with natural crust cover (%)	95				40		45		91	95

[a] In Sweden, payments are given to semi-natural pastures based on a classification into 'general values' or 'specific biological and cultural values'.
[b] Endangered domestic animal breed.
[c] Animal unit (AU) per hectare.
[d] Ton fat and protein corrected milk (FPCM) per year.
[e] Ton carcass weight (CW) per year.
[f] Percent of diet in dry matter (DM).

farms. The feed consists mostly of forage produced on-farm. This farm will be referred to as Suckler and dairy calves A.

4. *Dairy systems*

Two farms specialize in milk, but also produce some beef from culled dairy cows. The calves not used as replacement heifers are sold to other farms for fattening. The feed consists of forages, cereals, and other feed crops produced on-farm, plus concentrates. These farms will be referred to as Dairy A and B.

5. *Dairy and dairy calves system*

One farm, in addition to producing milk and beef from culled cows, also fattens calves not used as replacement heifers. The feed mostly consists of forages grown on-farm and some concentrates. This farm will be referred to as Dairy and dairy calves A.

Only total yearly feed consumption on-farm was known, so feed intake per animal was estimated based on gross energy (GE) requirements in animals, GE content in feed, and farmer-estimated total feed consumption (von Greyerz, 2021). Data on GE content in feed came from IPCC (2019b). GE requirements were calculated using IPCC (2019b) tier 2 separately for calves, replacement heifers, dairy cows, dry cows, breeding bulls, suckler cows, heifers for meat, and steers for meat, including requirements for maintenance, growth, activity, lactation, and pregnancy. The calculations were based on body weight, mature weight, weight gain, amount of milk produced, fat content in milk, and fraction of digestible energy in feed, using farm-specific parameters. When fat content in milk was unknown (for suckler cows), it was set to that in fat- and protein-corrected milk (FPCM), *i.e.*, 4% fat and 3.3% protein (International Dairy Federation, 2015) and with amount of milk produced according to Swedish Environmental Protection Agency (2019). Constants for maintenance energy, activity energy, and energy for growth were set according to IPCC (2019b), considering farm and bovine characteristics. Feed digestibility was calculated from reported amount of digestible energy in feeds (Swedish University of Agricultural Sciences, n.d). During the grazing period, dairy cows were assumed to consume 50% of their forage from grazing, based on Spörndly and Kumm (2010), while other cattle did not consume any other feed. Concentrate and cereal fraction in total feed intake was assumed to be similar for all animals on the farm, except for dry cows that were assumed to only eat forage, unless otherwise stated by the farmer (SM, Table S1). Forage fraction was then adjusted to match the required feed intake, considering the animal's energy needs. Feeding losses were assumed to be 3% for all animals except dairy cows, for which losses of 10% and 5% for forages and concentrates, respectively, were assumed, following Hessle et al. (2017).

Since herds can differ between years, *e.g.*, if the farm buys (or slaughters) more animals, a herd in equilibrium typical of each farm was used to calculate the carbon footprint, following von Greyerz (2021). The number of suckler cows and dairy cows were therefore held constant. The replacement rate was 20% for suckler cows (Cederberg, 2009), unless otherwise stated by the farmer (SM, Table S1), while for dairy cows the replacement rate was set to the number of cows slaughtered (reported by the farmer). The number of replacement heifers was set to equal the number of cows replaced. Each cow was assumed to give birth to one calf per year, unless otherwise stated by the farmer (SM, Table S1). Calf mortality and number of calves bought in were both set to the number reported by the farmer. The climate impact of bought-in dairy calves was based on calf weight and climate impact per kg dairy calf weight (Moberg et al., 2019). The climate impact of bought-in suckler calves was set to the impact from one suckler calf in one year and the impact of the growing calf based on calf age when bought in, with impacts taken from Moberg et al. (2019). No allocation to non-provisioning ES was made for these impacts, which might underestimate the effect of the allocation method for farms buying calves, depending on the calves' and the mother animals' contribution to ES and impact. Steer:heifer ratio for beef animals was set to that reported by the farmer for the study year, as was the mortality rate. When live-weight (LW) or carcass-weight (CW) was not stated by the farmer, the LW:CW ratio was set to 1:0.5, based on Strid et al. (2014).

2.2. System boundaries and functional unit

The functional unit (FU), *i.e.*, the quantitative reference unit for the system functions, chosen was 1 kg carcass weight (CW) for beef and 1 kg

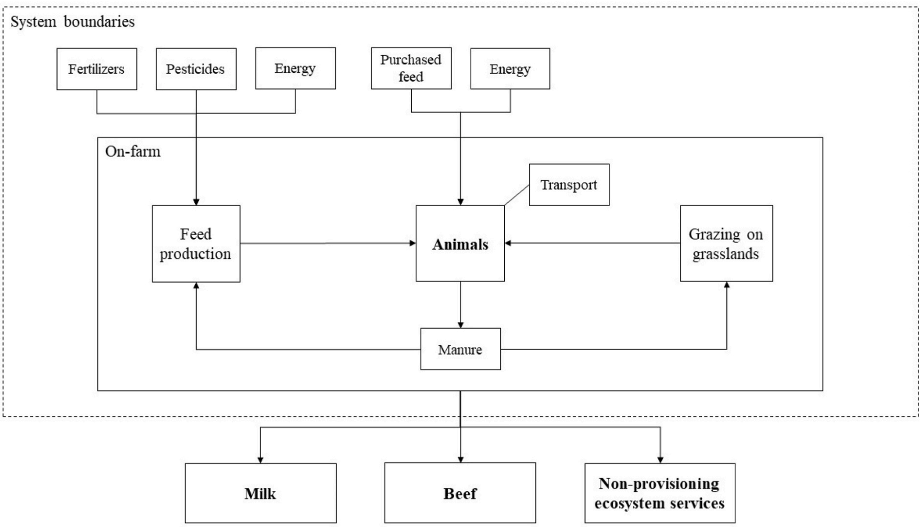

Fig. 1. System boundaries with inputs, outputs, and emission sources included in the analysis.

FPCM for milk. Processes from "cradle-to-gate", *i.e.,* until animals and milk leave the farm, were included in the system boundaries (Fig. 1). Emissions considered were: CH_4 from enteric fermentation, CH_4 and nitrous oxide (N_2O) from manure management, N_2O from grazing land due to manure deposited by grazing animals, and carbon dioxide (CO_2), CH_4, and N_2O from feed production, transport, on-farm energy use and purchased products. Emissions and sequestration of CO_2 from soil carbon stock changes caused by feed production were included for feed produced on-farm and for purchased feed. For feed produced on-farm, emissions associated with land use change were excluded, since the farms had not altered land use management substantially in the previous 20 years (IPCC, 2019a). For purchased feed, land use change was included for soy produced outside of Europe. For feeds produced in Europe, land use changes was excluded (Pendrill et al., 2020). Processes post-farm gate, *i.e.*, slaughter, processing, packaging, and transport, were assumed to be similar for all systems and therefore not included in the system boundaries. Capital goods were also excluded since it has been shown that these make minor contributions to climate impacts for agricultural products (Frischknecht et al., 2007).

2.3. Climate impact

The environmental impact category considered was climate impact, using CO_2e. For conversion to CO_2e, global warming potential in a 100-year perspective (GWP_{100}) with climate-carbon feedbacks (1 for CO_2, 34 for CH_4, 298 for N_2O), was used (IPCC, 2013). For more details on these calculations, see von Greyerz (2021) and supplementary materials. When assessing the environmental performance of different livestock systems it is important to consider a wide range of impact categories (van der Werf et al., 2020) to allow for a fair comparison and avoid pollution swapping. However, since the aim here was to study the influence of the allocation method, only one impact category was included since the allocation factors would be the same for all impact categories and therefore also the relative change of the impact.

Emissions of CH_4 from enteric fermentation were quantified with the tier 2 approach from IPCC (2019b) (SM, Table S2) from GE intake and a CH_4 conversion factor (Y_m) set to 6.3%, based on fraction of digestible energy in feed according to IPCC (2019b).

Emissions of CH_4 from manure management were quantified with the tier 2 approach from IPCC (2019b) (SM, Table S2) from volatile solids excreted by livestock and factors for the maximum CH_4-producing capacity and CH_4 conversion of manure taken from IPCC (2019b). Amount of volatile solids excreted was estimated from GE, ash fraction in feed, urine energy fraction in GE, and digestible energy fraction in feed, where ash fraction in feed was approximated with feed estimations and ash content from Swedish University of Agricultural Sciences (n.d), and urine energy fraction was based on IPCC (2019b).

Emissions of N_2O from manure management were quantified with the tier 2 approach from IPCC, (2019b) (SM, Table S2), using nitrogen (N) excretion from bovines calculated with IPCC, (2019b) tier 1 values, including direct and indirect emissions, the latter caused by N volatilization primarily as ammonia and nitrogen oxides and leaching. N excretion was estimated with IPCC (2019b) tier 1 from N intake and N retained in the animal, calculated using GE, protein fraction in feed, weight gain, and net energy for growth. Protein fraction in feed was calculated from feed estimations and protein contents from Swedish University of Agricultural Sciences (n.d).

Soils used for crop production generate direct and indirect emissions of N_2O (SM, Table S2), the latter caused by N volatilization primarily as ammonia and nitrogen oxides and leaching, from N added with fertilizer, manure, and crop residues. These emissions were calculated with the tier 2 approach from IPCC (2019c) from amount of added N, emissions factors for different amendments in wet climates from IPCC (2019c), and fractions of N volatilized and leached from IPCC (2019c), following the Swedish national inventory report 2019 (Swedish Environmental Protection Agency, 2019). Emissions from organic soils were not considered and all soils were treated as mineral soils. Nitrogen added with crop residues was calculated according to IPCC (2019c) tier 1, from yield, fraction of residues left in the field, and proportion of crops renewed annually, using values of above-ground residues:yield ratio, root-biomass:shoot-biomass ratio, and N content in residues from Andrist Rangel et al. (2016) and IPCC (2019c), also following the Swedish national inventory report 2019 (Swedish Environmental Protection Agency, 2019). Nitrogen added with synthetic fertilizers was calculated from N content in fertilizers and fertilizer use reported by farmers. Nitrogen from organic amendments and manure was calculated based on amounts reported by farmers and N content in similar amendments reported by Cool Farm Alliance (2019). Pastures also generate N_2O emissions when grazed, from manure deposited by grazing animals. Amount of N added to pastures with manure was estimated based on fraction of grazing period spent outside, estimated N excretion, and grazing period length reported by farmers. For farms where animals have outdoor access year-round, the manure was assumed to be collected and stored during winter (approximated as three months). For dairy cows, the fraction of the day spent outside was set to 70% (Wredle et al., n.d) when unknown. Nitrogen added to soils by grazing animals was calculated from estimated dry matter (DM) intake from grazing.

Emissions of GHG from energy use in barns and from feed production were calculated using emission factors for different energy sources from Gode et al. (2011). Emissions from transport were approximated following Kannan et al. (2016), based on vehicle and trailer weight, fuel consumption, and total weight of transported animals, choosing vehicle and trailer sizes similar to those on-farm.

Soil carbon stock changes were estimated for cropland using the Introductory Carbon Balance Model (ICBM) (Andrén et al., 2004), which estimates soil organic carbon content in topsoil on a yearly basis using initial soil carbon content and annual carbon input. A more detailed description is given in supplementary materials.

2.4. Valuable outputs and allocation

Valuable outputs considered were sold beef, milk and calves as well as non-provisioning ES. Cattle also generate manure, but since it was exclusively used in production of feed on-farm, and therefore did not leave the system, it was not considered an output. To distribute the climate impact between all outputs considered, economic allocation was used. Economic allocation excluding non-provisioning ES, *i.e.,* only considering beef, calves, and milk, was performed for comparison. For sold beef and milk, the economic value was calculated from amount sold and conventional producer prices in Sweden (2016) for both conventional and organic farms (Table 2). The reason why allocation between non-provisioning ES and food (milk/beef) was based on the conventional price, also for the organic farmers selling their products with a premium price, was that we considered the conventional price to best

Table 2
Producer prices for milk and bovines in Sweden (2016), used for allocation. Values converted from Swedish krona (SEK) to Euro (EUR) with conversion rate 10:1.

	Average price
Milk[a)] (EUR/kg)	0.31
Cattle sold to other farms[b)] (EUR/kg LW)	Calves: 2.8
	Heifers: 2.3
Cattle sold to slaughter[c)] (EUR/kg CW)	Culled cows: 4.0
	Young bulls: 4.3
	Heifers: 4.2
	Steer: 4.3

[a] Euro (EUR) per kg from Swedish Board of Agriculture, (n.d.a).
[b] EUR per kg live weight (LW) estimated from average for dairy breed from HKScan (n.d.).
[c] EUR per kg carcass weight (CW) estimated from average for class R3 for bulls and O3 for others from Swedish Board of Agriculture (n.d.b).

Table 3
Payments through the Swedish rural development program (2020 values) (Swedish Board of Agriculture, 2020) used for valuing ecosystem services in this study.

Payment	Description	Value
Maintenance of semi-natural pastures[a)]	Grazing of semi-natural grasslands	General value: 100 EUR/ha Specific values: 280 EUR/ha
Keeping of endangered domestic animal breeds[b)]	Breeding of endangered domestic animal breeds	145 EUR/AU
Farming on areas of natural constraints, ANC[c)]	Farming on areas with natural constraints, for pastures with specific values and crops.	Pastures with specific values: 100 EUR/ha Crops: 25–540 EUR/ha
Organic farming[d)]	For organic animal farming with organic cultivated land and/or semi-natural grasslands. Also for organic crops.	Animal units: 160 EUR/AU Grain, oilseed crops, and protein crops: 150 EUR/ha
Ley production	Production of leys in areas without natural constraints.	50 EUR/ha

[a] Euro (EUR) per ha.
[b] Value for cattle per animal unit (AU).
[c] Support for crops depending on AU per ha and location. Payments for pastures with specific values in addition to payment for maintenance of pastures.
[d] Given to organically farmed crops. If the farm also has animals, additional payments are given. Per AU, the farm must have 1 ha of organically farmed cropland or 2 ha of semi-natural pasture.

reflect the value for the physical food product itself (assuming equivalent quality of the food items). The added premium price for organic farming may in part reflect a value consumers are willing to pay for diverse public goods including non-provisioning ES.

As a proxy for the non-provisioning ES provided by the farms, payments through agri-environmental schemes under the EU Common Agricultural Policy (CAP) (European Parliament and the Council 1305/2013) was used, as done previously by Ripoll-Bosch et al. (2013), Kiefer et al. (2015), and Bragaglio et al. (2020). In this study, agri-environmental payments through the Swedish Rural Development Program (RDP) 2014–2020 associated with the studied cattle production were used (Table 3). The RDP specifies a need to restore, preserve, and enhance ecosystems related to agriculture and forestry, divided into three focus areas; biodiversity restoration, preservation and enhancement; water management; and soil erosion and management. For ES

Table 4
Grouping of payments used in economic allocation, where group 1 comprises payments directly connected to animal rearing, group 2 also includes payments for organic farming tied to livestock production, and group 3 also includes payments given for feed production.

Group 1	Group 2	Group 3
• Maintenance of semi-natural pastures • Keeping of endangered domestic animal breeds • Maintenance of semi-natural pastures with special values in areas of natural constraints	• Maintenance of semi-natural pastures • Keeping of endangered domestic animal breeds • Maintenance of semi-natural pastures with special values in areas of natural constraints • Organic farming (animal husbandry)	• Maintenance of semi-natural pastures • Keeping of endangered domestic animal breeds • Maintenance of semi-natural pastures with special values in areas of natural constraints • Organic farming (animal husbandry) • Organic farming (feed production) • Feed production in areas of natural constraints • Ley production

provided by ruminant production in these focus areas, farmers can receive payments for maintenance of pastures and keeping endangered domestic animal breeds (Swedish Board of Agriculture, 2020). In Sweden, payments are given to semi-natural pastures based on a classification into 'general values' or 'specific biological and cultural values' receiving higher payments (Swedish Board of Agriculture, 2020, 2021). Arable and livestock farms maintaining pastures with special values in ANC can also receive payments for contributing to the focus areas (Swedish Board of Agriculture, 2020). Land in ANC risk being abandoned by farmers (Hagyo et al., 2015), which poses risks to ES delivery. According to Hagyo et al. (2015), ANC generally have lower capacity to produce foods, but higher capacity to contribute positively to other ES (*e.g.*, habitat maintenance, pollination, recreation) than areas with more favorable conditions for agriculture. Farmers growing grass-clover leys can receive payments even when not located in an ANC. For contributions to the focus areas, payments are also made for organically farmed crops, with an additional payment for organically farmed animals (Swedish Board of Agriculture, 2020). Compared with conventional systems, organic farming can positively contribute to several ES, *e.g.*, increased biodiversity (Tuck et al., 2014), soil fertility and soil physical properties (Reeve et al., 2016), and improved water quality (Sivaranjani and Rakshit, 2019). Since organic management practices vary, the magnitude of the effect differs between organism groups and landscapes (Bengtsson et al., 2005).

For the economic allocation, the payments were divided into three groups depending on the connection to animal production (Table 4). Group 1 comprised payments directly connected to animal rearing *e.g.* maintenance of semi-natural pastures. Group 2, in addition to the payments in group 1, included payments for organic farming tied to livestock production which also are affected by agricultural land and Group 3 also included payments given for feed production.

3. Results and discussion

3.1. Climate impact without considering non-provisioning ES

The climate impact of beef and milk from the 10 farms when only considering beef, milk, and surplus calves as outputs are shown in Figs. 2 and 3 respectively. For beef, emissions were 13–36 kg CO_2e per kg CW excluding soil carbon stock changes and 16–39 kg CO_2e per kg CW when soil carbon stock changes were included. For milk, emissions were 0.76–1.2 kg CO_2e per kg FPCM excluding soil carbon stock changes and 0.66–1.1 kg CO_2e per kg FPCM when soil carbon stock changes were included. The higher emissions after including carbon stock changes are an effect of soils loosing carbon.

Beef from suckler farms (Suckler A-D) had the highest climate impact, followed by beef from the farm with both suckler and dairy calves (Suckler and dairy calves A). Meat from suckler herds generally has a higher climate impact than meat from dairy herds, as the emissions by the suckler cows are entirely allocated to the beef produced in suckler systems as these do not produce any milk for the market (de Vries et al., 2015). Animals on suckler farms (Suckler A-D) also had lower growth rates and higher slaughter age than animals on dairy farms (Dairy calves A and B), which increased the climate impact as more CH_4 from enteric fermentation was produced during the animal's lifetime and more feed needed to be produced. This confirms previous findings that beef produced on extensive farms commonly has a higher climate impact than beef from intensive farms (Ogino et al., 2016; Bragaglio et al., 2018). Previous research has shown that even with alternative allocation methods, including system expansion, most of the climate impact of dairy systems is attributed to the milk (Cederberg and Stadig, 2003; Baldini et al., 2017). The farm producing milk and fattened calves for beef (Dairy and dairy calves A) had a higher climate impact for beef and milk than the other dairy farms, owing to extensive fattening of surplus calves for beef and lower milk yield. This farm also had a higher fraction of climate impact allocated to beef than the other dairy farms (Dairy A,

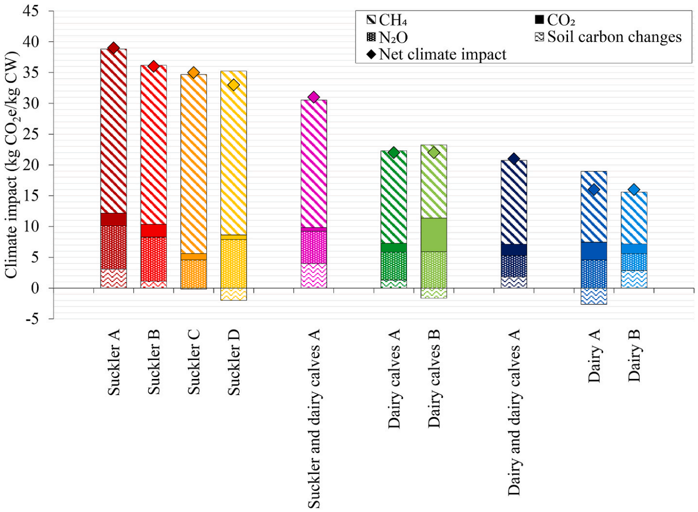

Fig. 2. Climate impact in kg carbon dioxide equivalents (CO_2e) per kg carcass-weight (CW) when only considering beef, milk, and surplus calves as valuable outputs. Impacts are subdivided into carbon dioxide (CO_2) from energy use, land use change and transport, methane (CH_4) mainly from enteric fermentation and manure management, nitrous oxide (N_2O) mainly from manure management and emissions from soils caused by N additions (e.g. crop residues), and CO_2 emissions or sequestration from carbon stock changes. Net climate impacts are also shown.

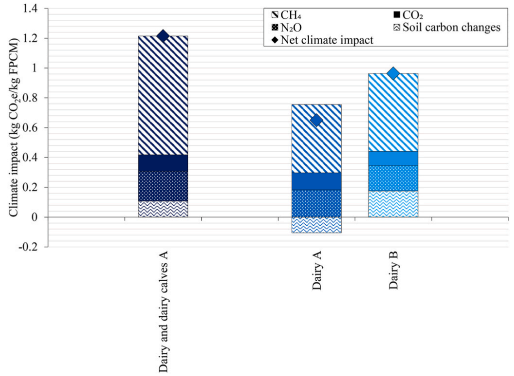

Fig. 3. Climate impact in kg carbon dioxide equivalents (CO_2e) per kg fat- and protein-corrected milk (FPCM) from dairy farms when only considering beef, milk, and surplus calves as valuable outputs. Impacts are subdivided into carbon dioxide (CO_2) from energy use, land use change and transport, methane (CH_4) mainly from enteric fermentation and manure management, nitrous oxide (N_2O) emissions mainly from manure management and emissions from soils caused by N additions (e.g. crop residues), and CO_2 emissions or sequestration from carbon stock changes. Net climate impacts are also shown.

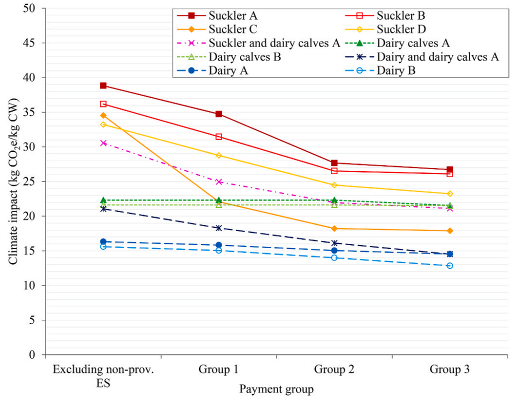

Fig. 4. Climate impact of beef in kg carbon dioxide equivalents (CO_2e) per kg carcass weight (CW) when excluding non-provisioning ecosystem services (non-prov. ES), *i.e.* only including beef and milk for allocation, and when including ecosystem services for allocation using: payment group 1 (payments directly connected to animal rearing), group 2 (also including payments for organic farming tied to livestock production), and group 3 (also including payments for feed production).

B), owing to its higher income from beef due to the value added by including the fattening phase on-farm rather than selling live animals for fattening elsewhere.

Including soil carbon stock changes led to higher estimated climate impact for four of the farms (Suckler A, Suckler B, Suckler and dairy calves A, Dairy B), mostly due to high initial carbon stocks in topsoil in the area where the farms were located, resulting in carbon losses. Soils on three other farms (Suckler D, Dairy calves A, Dairy A) sequestered carbon instead, due to lower initial carbon stocks in those areas. It should be noted that modelling soil carbon changes is associated with large uncertainties, especially for cropping systems consisting of high proportion of leys for which the yield level is difficult to estimate. According to a soil monitoring program in Sweden, decadal carbon sequestration on beef and in particular dairy farms has been substantial but changes in soil organic carbon also show a high spatial and temporal variation between farms (Henryson et al., 2020).

3.2. Climate impact when considering ES for different farm types

When also considering non-provisioning ES provided by the farms, as captured by payment-schemes, the difference in climate impact of beef between farms were smaller, 13–27 kg CO_2e per kg CW (instead of 16–39 kg CO_2e per kg CW), allocating <1–48% of the climate impact to

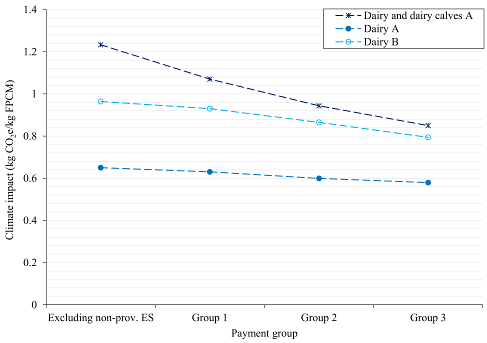

Fig. 5. Climate impact of milk in kg carbon dioxide equivalents (CO_2e) per kg fat- and protein-corrected milk (FPCM) when excluding non-provisioning ecosystem services (non-prov. ES) for allocation and when including ecosystem services for allocation using: payment group 1 (payments directly connected to animal rearing), group 2 (also including payments for organic farming tied to livestock production), and group 3 (also including payments for feed production).

non-provisioning ES (Fig. 4). The climate impact of milk also decreased when considering ES in allocation, from 0.65 to 1.2 to 0.58–0.85 kg CO_2e per kg FPCM, allocating 11–31% of the climate impact to other ES (Fig. 5).

Including non-provisioning ES as an output had the largest effect on the climate impact of beef from suckler farms (Suckler A-D, Suckler and dairy calves A). When group 3 was used for allocation, *i.e.*, including all payments considered in this study, 10–17 kg CO_2e per kg CW were allocated to non-provisioning ES, corresponding to 23–48% of the climate impact. The suckler farms used extensive management methods with low amounts of inputs and high reliance on pasture, resulting in lower growth rates and thus higher slaughter ages. The suckler farms hence produced less provisioning ES in terms of food per ha, but contributed more positively to other ES. Therefore, the allocation method affected the climate impact for beef more than for farms breeding dairy calves more intensively and dairy farms producing both milk and beef. Bragaglio et al. (2020) also found that extensive farms were most affected when including non-provisioning ES but the relative effect was smaller, mainly because the extensive farms they studied used an indoor fattening phase and had higher growth rates than the extensive farms assessed in this study.

Compared with the suckler farms, the specialist dairy farms (Dairy A and B) had a smaller shift in climate impact for beef, with 2–3 kg CO_2 per kg CW allocated to the non-provisioning ES, corresponding to 11–18%. The relative shift for milk was the same, corresponding to 0.17 and 0.07 kg CO_2e per kg FPCM, respectively. In Kiefer et al. (2015), 1–29% of the climate impact from milk was allocated to non-provisioning ES. For their cluster of farms most similar to Dairy A and B (pasture-based with similar milk yields and breed), 8% was allocated to non-provisioning ES, a somewhat lower fraction than for Dairy A and B. However, the payments for managing grasslands were generally lower in Kiefer et al. (2015) (50–120 EUR/ha) compared with this study (100 and 280 EUR/ha). Dairy A and B generated an economic value for non-provisioning ES per ha of the same magnitude as the suckler farms, suggesting a similar positive contribution to ES per land area used. However, since dairy farms generate more income per ha and animal from foods produced (due to the production of milk), allocation factors were less affected by the income from other ES, resulting in a smaller shift. This allocation method should therefore be used with caution when comparing dairy farms with beef farms. In addition, for the farms producing both beef and milk, the allocation method by definition gave the same relative shift for milk and beef, suggesting that milk and beef production contributed equally to non-provisioning ES. However, this may not reflect reality, since on farms producing milk and rearing surplus calves for beef, the latter can potentially contribute more to ES by *e.g.*, longer grazing periods.

The climate impact of beef from Suckler C was most affected by the allocation. Suckler C had the lowest animal density, and therefore provided the least amount of beef per ha. Instead, larger areas of pasture were managed per animal, resulting in more positive contributions to ES per kg CW. Suckler C also received payments for rearing endangered domestic breeds. Similarly, Bragaglio et al. (2020) found that the most extensive system using native breeds was most affected by this allocation method, shifting impacts from the beef to the other services.

Overall, the shift in climate impact to non-provisioning ES (<1–48%) was of the same magnitude as reported by Bragaglio et al. (2020) (0–43%). When comparing dairy farms only, the shift (11–31%) was similar to that in Kiefer et al. (2015) (1–29%). The differences between the studies were partly caused by differences in production, but also by including different payments. Since it is unclear which payments are directly connected to animal rearing, the results depend on the decision of which payments to include. Since the payments vary between countries, the results also reflects nation specific factors, e.g. valuation of ES, politics and finance (Ecorys et al., 2017), making it difficult to compare results across countries. Moreover, the method is sensitive to changes in payments over time, whereby the assumed value of non-provisioning ES also change over time, making it difficult to compare results from different years (Kiefer et al., 2015).

3.3. Variation due to payment schemes included

To analyze the effect of including different payments, they were grouped here according to their level of connection with livestock production. Overall, group 1 payments (maintenance of semi-natural pastures, endangered domestic animal breeds, and maintenance of semi-natural pastures with 'special values' within an ANC) gave a shift of up to 12 kg CO_2e (36%), group 2 (also including payments for organic animal farming) shifted another 0–7 CO_2e (0-18%) from food provisioning to other ES, and group 3 (also including payments for feed production), shifted an additional 0–2 kg CO_2e (0-8%). This indicates that payment schemes that are more directly connected to the ruminant production systems make the largest positive contribution to ES according to how these are valued by society, which is however a result of policy decisions conflated by multiple priorities besides supporting non-provisioning ES (Ecorys et al., 2017). The ANC payments depended on location and animal density, with most farms receiving lower payments for this than for management of pastures. The payment for ley production was lowest of all payments considered. The payments per ha for organic farming of cereals and oilseed crops were higher than for pastures with general values but, since most of the on-farm produced feed consisted of ley, the payment for organic farming of cereals and oilseed crops for feed barely affected the allocations. This resulted in lower payments from feed production than from payments directly associated with rearing of the animals, therefore affecting the allocation factor the least. Suckler A was most affected by group 2 payments, owing to its higher animal density and smaller area of pasture managed than on the other farms. The dairy farms had a smaller area of semi-natural pastures per economic value of products (meat and milk) than the suckler farms, resulting in a smaller effect from group 1 payments. For Dairy and dairy calves A, group 3 payments had a larger effect on allocation than on the other farms, explained by this farm being located within an ANC with higher payments. This indicates that the support for feed production can be important for farms producing their own feed if located in a specific ANC.

In this study, ES connected to the focus areas in the Swedish RDP were considered, *i.e.*, "Biodiversity restoration, preservation and enhancement", "Water management" and "Soil erosion and soil management", as the payment schemes provide an economic value indicating what society is currently paying. Non-material services such as physical and psychological experiences, are difficult to value economically, but some may be indirectly captured through payments for *e.g.*, maintenance of semi-natural pastures that are associated with cultural values (Karlsson et al., 2022).

A potential alternative to economic allocation based on payment schemes is to use a scoring method to score the (capacity for) delivery of provisioning and other ES from the system under study and base allocation on these scores, as suggested by (Boone et al., 2019). This could avoid assumptions on how well payment schemes capture the supply of ES and allow for the inclusion of more ES (not covered by payment schemes) but would necessitate some procedure to weigh the importance of different ES. Boone et al. (2019) assumed an equal weight on provisioning and regulating ES, which is unlikely to accurately reflect how different ES are valued in society. This valuation could however be done with e.g. choice modelling where different stakeholders are asked to value different ES (Faccioni et al., 2019). Deriving allocation factors this way would however be sensitive to which stakeholders are included in choice experiments (Bernués et al., 2014) and deriving transparent, non-context specific and generalizable factors may be hard. Using income from payment schemes avoids this by assuming that the size of these payments reflects society's prioritization between different ES.

3.4. Using climate impact values for beef with emissions allocated to ES

Results from LCAs are used as decision support in a range of applications in the food system, including labeling for consumer communication, monitoring of environmental impacts in food production for policy development and evaluation, and guiding environmental improvements of industry's food production (Notarnicola et al., 2017). Food companies are increasingly using climate impacts as part of consumer communication. For example, the Swedish online retailer Mat.se labels 3000 food products with their carbon footprint[1] and ICA, the largest retailer in Sweden, provides its loyalty card holders with a monthly summary of aggregated emissions from their food purchases.[2] These measures are intended to act as drivers in reducing GHG emissions through influencing consumer choice, *i.e.,* consumers choosing products with lower climate impacts, and through improvements in production, *i.e.*, food producers (farmers and food industry) lowering emissions through efficiency improvements, technological advances, reduced waste, or changes in ingredients in composite foods. More intensive beef production systems tend to have lower climate impacts per kg of meat than extensive, multifunctional systems. Therefore, there is a risk of pushing production systems towards more intensive production when non-provisioning ES are not considered in the climate impact calculations. This would neglect important values that multifunctional ruminant systems could deliver. Including non-provisioning ES in climate impact calculations, as done in this study, can reduce this risk. Another option could be to label or monitor the outcomes for non-provisioning ES alongside climate impacts and present several environmental indicators for each food product. For example, the Swedish retailer COOP provides sustainability declarations for some products based on 10 sustainability indicators[3] (climate, biodiversity, soil fertility, water, pesticide use, eutrophication, animal welfare and use of antibiotics, working conditions, local community, rule of law and tractability) in a 'spider's web' diagram. This covers a greater range of sustainability aspects, which is important for foods considering the potential trade-offs. However, it also leaves the consumer to weigh these aspects, increasing the complexity in consumer communication (Ströbele and Lützkendorf, 2019). Considering non-provisioning ES as an output of the system and allocating some of the climate impact to these ES might be a more straightforward solution that has the simplicity of just one indicator, climate impact, while considering the benefits of multifunctional systems. However, use of this method in practice can be challenging as impacts can vary over time and country, due to changes and differences in payment schemes, making it difficult to fairly compare products. More research is needed into the practical use of ES-corrected climate impact.

An actor in the food system that could benefit from including ES in climate impact assessments is the Swedish organization KRAV, (that develops standards for organic certification, in addition to the EU regulations) which from 2022 requires all farms larger than 200 ha to calculate and report their climate impact (KRAV n. d.). At the time of writing it is unclear how KRAV will use the climate impact data. If used to compare farms in terms of climate impact per kg food produced, in order to incentivize reductions in emissions by certified farmers, it could lead to intensification of organic farms and compromised animal welfare and biodiversity outcomes (Röös et al., 2018). Considering non-provisioning ES in the climate impact calculations could alleviate that risk, as delivering more ES would also be a way to improve the climate impact value.

In all applications of ES-corrected climate impacts, it is important to acknowledge that emissions will not disappear, but will only be shifted from beef or milk to other ES provided. To reach climate targets, very drastic cuts in emissions are needed, including in food systems and agriculture (Clark et al., 2020). Thus, when impacts are shifted from foods to other ES, reducing emissions from provision of these ES must not be forgotten. For example, semi-natural pastures can be managed for biodiversity conservation in more or less climate-impacting ways. According to Röös et al. (2016), managing these pastures with suckler herds instead of animals from dairy production is more climate-efficient per ha managed land as it requires fewer animals in total (since suckler cows have longer grazing periods than dairy cows). This was confirmed in the present study, where Suckler C delivered non-provisioning ES at a much lower total climate cost per ha than the other farms, as fewer animals grazed a larger area and animal feed intake was dominated by grazed biomass. Since managing emissions from ES might be the responsibility of policy makers for the food system, rather than farmers or consumers, allocating emissions to the additional ES could make this responsibility more transparent and explicit.

4. Conclusions

Including non-provisioning ES in addition to food provisioning services when attributing the climate impact from ruminant systems had a large effect and was affected by different ways of coupling ES to livestock through payment schemes. Including payments for ES most directly associated with animals (here represented by payments for management of pastures and endangered domestic breeds) had the largest effect on the climate impact, while ES related to feed production had a smaller effect. The magnitude of the effect from the different coupling approaches depended on animal density, location, and area of semi-natural grasslands, as an outcome of policy decisions on compensatory payments. Including non-provisioning ES in the allocation resulted in <1–48% and 11–31% of the climate impact being shifted from beef and milk, respectively, to other ES. Suckler farms were most affected, while dairy farms had a smaller shift owing to high production of milk. ES-corrected climate impact can potentially be useful as part of consumer communication or as a decision tool for policy makers and industry, reducing the risk of neglecting non-provisioning ES provided by ruminant production in a simpler way than using separate indicators. However, it is important to note that emissions do not disappear, but are only shifted from beef and milk to other ES.

1 https://www.mat.se/mat-klimat.

2 https://www.ica.se/buffe/artikel/mitt-klimatmal-info/.

3 https://www.coop.se/hallbarhet/hallbarhetsdeklaration/.

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.

Acknowledgments

This work was foremost supported by the Swedish University of Agricultural Sciences, but also the Uniseco project (an EU founded research project, https://uniseco-project.eu/) [Grant agreement number 773901], and Swedish Farmers' Foundation for Agricultural Research, [Grants R-18-26-136].

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.116400.

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Renewable Agriculture and Food Systems

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Research Paper

Cite this article: von Greyerz K, Ericsson N, Jardstedt M, Hessle A, Arvidsson Segerkvist K, Röös E (2024). Feeding straw to suckler cows spared land but did not decrease the climate impact of beef. *Renewable Agriculture and Food Systems* **39**, e36, 1–12. https://doi.org/10.1017/S1742170524000255

Received: 6 December 2023
Revised: 20 June 2024
Accepted: 2 September 2024

Keywords:
alternative feeds; cattle; greenhouse gas emissions; land use; multifunctionality; sustainable food

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Feeding straw to suckler cows spared land but did not decrease the climate impact of beef

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Abstract

Beef has a considerably higher climate impact than meat from monogastric animals and plant-based foods, due to methane emissions from enteric fermentation in ruminants. Animal feed production also contributes considerably to the climate impact, through carbon dioxide emissions from fossil fuel use and nitrous oxide emissions from soil. Despite this, ruminant animals can still be part of sustainable food systems, as they can produce human-edible food from coarse biomass unsuitable for human consumption (e.g., grass or straw), i.e., acting as 'upgraders'. Feeding ruminants on coarse biomass also reduces the need for cropland for feed production. Using cereal straw as indoor feed for suckler cows reduces their feed intake in winter, while increasing their intake of biomass on pasture during the grazing season. This study assessed the climate impact of producing 1 kg of beef (carcass weight), and of the farm as a whole, in a Swedish suckler-based system using a mixture of cereal straw and grass-clover silage as winter feed for suckler cows, compared with using only grass-clover silage (reference scenario). The rest of the feed remained unchanged. Replacing part of the grass-clover silage with straw meant that less cropland area was needed to grow feed. Two alternative scenarios for using this spared land were investigated: producing wheat for human consumption (straw-food) and conversion to pasture (straw-pasture). Effects on total food production were also calculated. Using a combination of cereal straw and grass-clover silage as winter feed for suckler cows was found to reduce the climate impact associated with feed production compared with using only grass-clover silage. However, this change in winter feed increased biomass intake on pasture during the grazing season and thus the grazed area, so total climate impact of beef per kg carcass weight, and of the farm as a whole, increased when the demand for more grazing area resulted in deforestation. With no deforestation, the climate impact was comparable to that of beef from suckler cows fed exclusively on grass-clover silage during winter. Therefore, upcycling of straw to meat had no notable effect on the climate impact, indicating that using residues as feed does not always entail a climate benefit. However, increased demand for pasture can have a direct benefit for biodiversity if more biologically rich semi-natural pastures are maintained or restored. Using the land spared through feeding straw instead of grass-clover silage for wheat production increase total food production from the system, with potential indirect climate benefits.

Introduction

Beef has a considerably higher climate impact than meat from monogastric animals and plant-based foods, through causing emissions of the greenhouse gases carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) (Clune, Crossin and Verghese, 2017; Moberg et al., 2019; Poore and Nemecek, 2018). The high climate impact is mainly due to methane emissions from enteric fermentation in the rumen of cattle during feed digestion and emission of greenhouse gases during the production of feed, mainly carbon dioxide from fossil fuels and nitrous oxide from soils. Additional emissions include nitrous oxide and methane from manure management and energy use, and nitrous oxide from excreta produced by animals during grazing (Hammar, Hansson and Röös, 2022; von Greyerz et al., 2023). Climate impacts from beef production are commonly quantified using a life cycle perspective, meaning that all impacts throughout the life cycle are considered, i.e., impacts from production of inputs such as fertilizers and fuels, on-farm activities such as feed production and animals, transport, slaughter, meat processing, final preparation, and waste management. For beef (and other animal products), a common simplification is to only include the climate impact up to farm gate (Hammar, Hansson and Röös, 2022; von Greyerz et al., 2023), since most of the impact occurs before that point (Moberg et al., 2019). Moreover, in studies of different on-farm management practices, impacts beyond the farm gate can be considered similar and hence do not influence comparisons. Swedish beef has an estimated climate impact of approximately 19 kg carbon dioxide equivalents (CO_2e) per kg of carcass weight at the farm gate (Ahlgren et al., 2022) which is similar to the global average of approximately 20 kg $CO_2e\ kg^{-1}$ of carcass weight (Clune, Crossin and Verghese, 2017). However, emissions vary substantially depending on how the beef is produced (Ahlgren et al., 2022; Clune, Crossin and Verghese, 2017; Poore and Nemecek, 2018). Thus, beef is a food item with considerable climate impact, but cattle systems add value to the food system in other ways and such cases can be considered multifunctional.

An important value provided by cattle is production of nutrient-dense foods from biomass unsuitable for human consumption (such as grass and straw), due to the ability of ruminants to digest cellulose-rich feeds (Karlsson, 2022). Using these feed types reduces food–feed competition and can also reduce the area of cropland needed to feed a growing global population (Van Zanten et al., 2018) and the climate impact from feed production (van Hal et al., 2019). In addition, grazing by cattle on semi-natural pastures with natural, cultural, and historical values plays a crucial role in promoting biodiversity by providing a habitat for endangered species (Eriksson, 2022). Unfortunately, these vital landscapes are undergoing rapid decline, primarily due to ceased grazing. In Sweden, the Prioritized Action Framework for Natura 2000 includes the goal of restoring 84,000 ha of semi-natural grassland by 2027 (Swedish Environmental Protection Agency, 2021b).

When calculating the climate impact of beef, commonly only the beef and milk produced are taken into account, but not other values that can be provided by beef production systems. Accounting for additional values, e.g., the grazing services that cattle provide, can affect the climate impact per kg beef. For example, von Greyerz et al. (2023) showed that including non-provisioning ecosystem services in the assessment can substantially reduce the climate impact per kg of beef.

The intensity of emissions from feed production varies depending on feed type and production system. The nutritional content of the feed also affects emissions from manure management and enteric fermentation in animals (IPCC, 2019a). This means that reducing emissions from feed production does not always lead to a lower climate impact per kg of beef or of the farm as a whole. Therefore, it is important to apply a life cycle perspective, i.e., to include all emission sources, when studying the climate impact from these systems. Methodological choices in life cycle assessment, such as how emissions from shared processes are allocated to different products (e.g., how emissions from cereal cultivation are divided between grain and straw), also influence the climate impact associated with feed production (Flysjö, Cederberg and Strid, 2008).

Beef is produced either as a dairy system by-product from culled cows, bull calves, and surplus heifer calves, or in suckler beef systems. Beef from dairy systems and suckler beef systems each comprises approximately half of Sweden's domestic beef supply (The Federation of Swedish Farmers, 2023). In suckler systems, calves stay with the dams until 6–8 months of age and are then weaned, finished, and slaughtered as young cattle. After weaning, male calves reared as bulls are usually reared indoors on a feed ration consisting of about 50% grass-clover silage and 50% grain and other types of concentrate, and slaughtered at approximately 18 months of age. However, male calves can also be reared as steers on forage and grazing, typically reaching a higher slaughter age (on average approximately 26 months). This is usually also the case for heifers (Ahlgren et al., 2022). Beef is also obtained from culled suckler cows.

Suckler cows typically graze during summer and are kept indoors and fed forage, mostly grass-clover silage, during winter (Ahlgren et al., 2022). However, *ad libitum* provision (a common management system) of early cut forage can lead to overfeeding. Using more fiber-rich forage such as cereal straw could reduce winter feed intake and thus environmental impact and feed costs (Jardstedt, 2019).

The aim of this study was to assess the climate impact replacing part of the grass-clover silage fed to suckler cows with straw during winter. The climate impact was assessed per kg beef (carcass weight) and for the farm as a whole. Effects on total food production at the farm level were also studied.

Methods

System description

This study compares two types of winter feed for suckler cows: one consisting of a combination of grass-clover silage and straw, and the other using only grass-clover silage. For the other cattle (bulls and heifers, respectively), the feed remained the same across scenarios. The study was based on modeling a theoretical farm of 33 ha cropland and a variable area of semi-natural pasture (depending on the increased amount of grazed grass as a result of the reduced intake associated with feeding of straw during winter) (Fig. 1). The farm was assumed to be located in the Götaland forest district in southern Sweden, producing beef in a typical Swedish integrated beef suckler system as described by Ahlgren et al. (2022).

It was assumed that the farm kept 30 suckler cows and borrowed two bulls for breeding from a neighboring farm. Each cow gave birth to one calf per year, weaned at 7 months of age, and half the calves were assumed to be bull calves. All bull calves and the heifer calves not used for replacement were assumed to be fattened to beef on-farm. Replacement and mortality rates were set according to Ahlgren et al. (2022) (Table 1). The spring-born calves grazed together with the suckler cows until weaning, after which bulls for beef were kept indoors until slaughter, while heifers also grazed for 5 months in the following summer before finishing. Cows and breeding bulls had an average yearly grazing period of 5.5 months. It was assumed that suckler cows and their calves grazed only semi-natural pasture while older growing animals also grazed ley aftermath. The animals were assumed to graze 1656 kg dry matter of forage per ha from the semi-natural pasture during the grazing season, based on reported average feed intake rates for different pasture types (Ahlgren et al., 2022).

Bulls and heifers were fed forage, cereals, legumes, and mineral supplements during the stable period (Table 1), using intakes from Ahlgren et al. (2022). Feed losses were accounted for by assuming that 11% of forages and 2% of other feeds were wasted after field to mouth. The weaned bull calves were fed grass-clover silage, cereals, and legumes whereas the weaned heifers were fed grass-clover silage and mineral supplements during winter. Breeding bulls were fed the same feed as finishing bulls until 15 months old. Afterward, they were fed whole-crop silage and mineral supplement during winter, and grazed during summer.

All feed was assumed to be produced on-farm, except for mineral supplements. Cropland and semi-natural pasture use in the scenarios was calculated based on livestock diet and crop yields (Fig. 1). Assumed yields, energy use, and nitrogen demand for cultivation are shown in Table 2. All cropping was assumed to

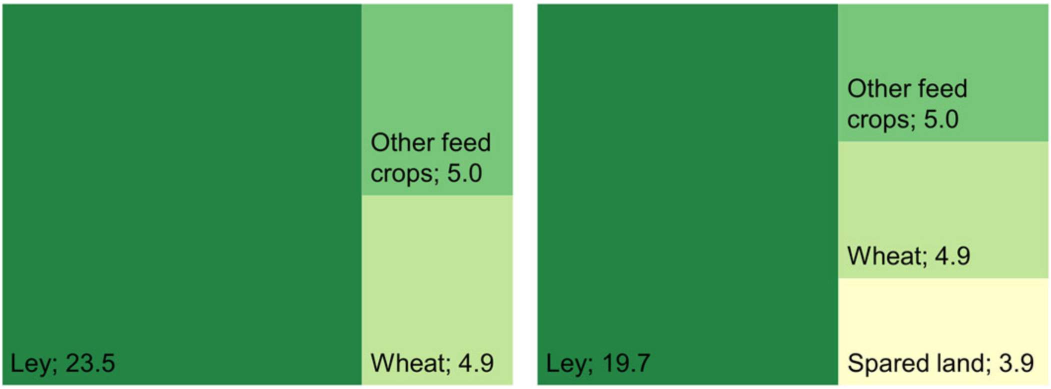

Figure 1. Cropland use (ha) in (left) the reference scenario and (right) the straw scenario.

Table 1. Production characteristics in the reference and straw scenarios

	Suckler cows					
	Reference scenario	Straw scenarios	Bulls—after weaning	Heifers—after weaning	Breeding bulls—from 16 months	Calves—before weaning
Slaughtered animals per year	5	5	14	9	0.5	
Age at slaughter (months)	79	79	15	24	53	
Carcass weight (kg)	375	375	360	315		
Mortality (%)			2.2	1.6		Bulls: 4 Heifers: 3.2
Average feed intake (kg dry matter day^{-1})						
Pasture herbage	10.6	12		7.2	11	4–7 months: 1.47
Whole crop silage					10.4	
Grass-clover silage	13	6.0	5.1	6.9		
Grass silage	16.5	16.7				
Straw		4.6				
Cereals			3.5			
Legumes			0.28			
Mineral feed	0.10	0.10	0.05	0.05	0.11	
Milk (from suckler cow) (kg day^{-1})						0–3 months: 10 4–7 months: 6.0

be conventional, using ley yields from Ahlgren et al. (2022) and standard yields for conventional cropping in the Götaland region in 2022 (Swedish Board of Agriculture, n.d.b.). Amount of straw available for harvest was approximated using the fraction of straw harvested for crops from Nilsson and Bernesson (2009), with similar biomass distribution as in Bertilsson and Nilsson (2020) (used for other calculations). Field losses from Andersson et al. (2022) were used for forages. The amount of nitrogen added to soil as synthetic fertilizer was calculated as the difference between total nitrogen demand (approximated with recommendations for fertilization; Andersson et al., 2022) and nitrogen from manure produced on-farm. Leys provide nitrogen for the next crop grown on the same field and the amount was assumed to be 40 kg nitrogen per hectare (ha) for mixed leys (grass and clover) and 15 kg for grass leys (Andersson et al., 2022). For energy use in cropping, data from Flysjö, Cederberg and Strid (2008) were used (Table 2). This included cultivation, drying, and ensiling.

The manure from suckler cows and heifers was assumed to be in the form of deep-bedded manure during the winter season. In the summer, all excretion took place on pasture. The manure

Table 2. Yield, nitrogen demand, and energy use for production of the different crops

	Yield (tons dry matter ha^{-1})	Nitrogen demand (kg ha^{-1})	Diesel (MJ ha^{-1})	Oil (MJ ha^{-1})	Electricity (MJ ha^{-1})
Ley, grass					
Year 1	2.5	80	1550		
Years 2–3	7.9	190	1550		
Ley, grass-clover mixture (20% clover)					
Year 1	2.5	60	1550		
Years 2–3	7.9	140	1550		
Winter wheat, grain	5.5	145	2998	2016	113
Winter wheat, straw	5.9				
Spring barley, grain	3.5	70	2808	792	77
Spring barley, straw	1.6				
Whole crop silage, barley	3.6	50	2836	792	77
Fava beans and peas	2.8		2808	1296	66

from the bulls was assumed to be in the form of slurry (Ahlgren et al., 2022).

The energy source used in animal houses was assumed to be electricity (Moberg et al., 2019) and the total amount of electricity used was set at 88 MWh yr^{-1} (Baky, Sundberg and Brown, 2010).

Reference scenario

From calving to grazing (70 days yr^{-1}), all cows were assumed to be fed grass silage. From housing to calving (120 days yr^{-1}), suckler cows were fed grass-clover silage. During summer grazing they were only feeding on the pastures (Fig. 2). Feed intakes from Ahlgren et al. (2022) and Jardstedt et al. (2020) were used (Table 1).

Straw scenarios

The feed for suckler cows were the same in both straw scenarios. From calving in spring through summer grazing until housing the feed types were the same as in the reference scenario, but with different levels of intake. From housing to calving (120 days yr^{-1}), the feed differed from the reference scenario. In the straw-feeding

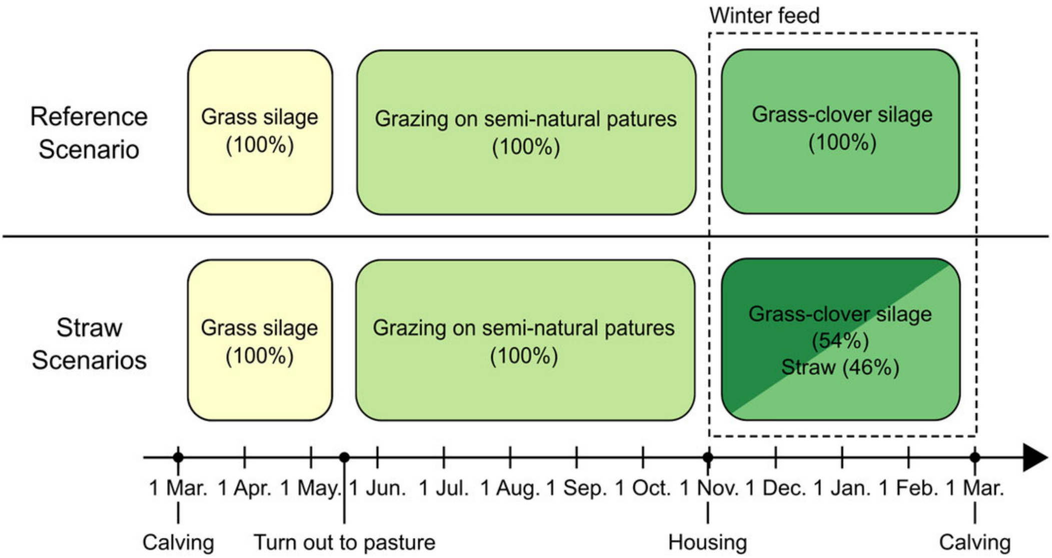

Figure 2. Feed rations fed during different periods of the year in the reference and straw scenarios.

scenarios, suckler cows were fed a combination of straw and grass-clover silage, replacing 56% of grass-clover silage fed to the suckler cows in the reference with straw during this period (Table 1; Jardstedt et al., 2020; Holmström 2022). The intake of straw and grass-clover silage was calculated using NorFor (n.d.). Grass silage from Jardstedt et al. (2020) was used, and the intake from grazing was calculated in accordance with Spörndly (2003), based on the pastures' chemical composition. For the other cattle (bulls and heifers), the feed rations where the same as in the reference scenario (Table 1). Feed rations for suckler cows during different periods of the year are shown in Figure 2.

By replacing parts of grass-clover silage with straw from cereal production (grown on-farm, see below), less cropland area was needed to grow feed. Two scenarios with different uses of this 'spared' land were assessed:

- Straw-food: suckler cows were fed a combination of straw and grass-clover silage as winter feed with the spared cropland area used to grow wheat for human consumption.
- Straw-pasture: suckler cows were fed a combination of straw and grass-clover silage as winter feed with the spared cropland area converted to pasture for cattle grazing.

Feed intake on grass was assumed to be higher for cows in the straw scenarios where the cows were partly fed with straw during the previous winter compared with cows fed only grass-clover silage (Table 1; Hessle 2022) since suckler cows eat more while on pasture when previous nutrient intake have been restricted by feeding on straw during winter. In the straw-food scenario, it was assumed that some previously abandoned semi-natural pastures needed to be restored, as the 'spared' cropland was used to produce additional wheat. Semi-natural pastures can be restored from different land types, but since most former pasture in Sweden is forested, deforestation will potentially occur. In the straw-pasture scenario, the spared cropland provided the extra permanent pasture needed. For this new pasture established on former cropland in the straw-pasture scenario, total yield (including grass not grazed) was calculated to be 3660 kg ha^{-1}, based on Ahlgren et al. (2022). Production characteristics for all scenarios are shown in Table 1.

Climate impact calculations

Processes up to the farm gate were considered in the climate impact assessment, including production of inputs (i.e., synthetic fertilizer, diesel, fuel oil, electricity, and mineral feed), feed production on-farm, methane emissions from animals, and manure management. Production and maintenance of buildings and machinery were excluded, as were manufacture of medicines, scouring agents, and other substances used in small volumes. Transport of inputs makes only a small contribution to emissions from these systems (Moberg et al., 2019), and was therefore also excluded. System boundaries are shown in Figure 3.

The climate impact results were presented for the farm as a whole and per kg of carcass weight, where carcass weight was assumed to be 53–61% of live weight (Ahlgren et al., 2022). Beef production generates a range of beef meat cuts and also by-products in the form of offal and blood. The entire climate impact from beef production was allocated to the carcass, even

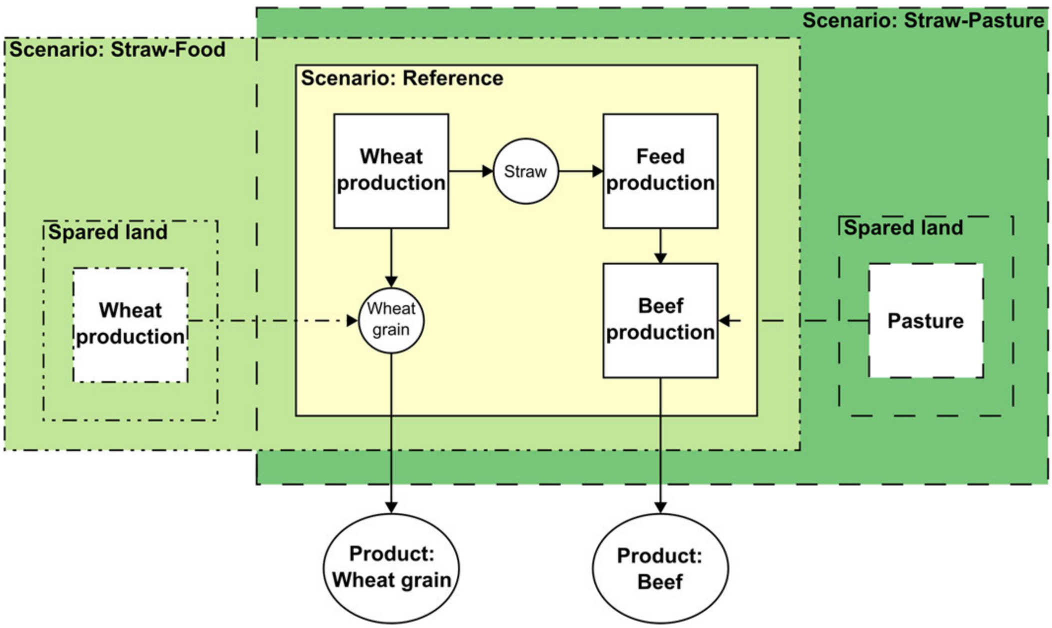

Figure 3. System boundaries applied for the different scenarios, with included processes and products.

though some of the by-products (skin, entrails, blood, and bones) are used in other applications. The economic value of the by-products is substantially lower than that of beef, meaning that if economic allocation would have been used only a small share of emissions would have been allocated to the by-products. Thereof, this simplification had no substantial effect on the comparison between the two systems.

The farm also produced wheat grain. All emissions from wheat cultivation are usually allocated to the grain, considering the straw as a by-product when used for bedding (Berglund et al., 2013) or a waste product when returned to the soil (Moberg et al., 2019). The same approach was used in this study, to reflect the ability of cattle to upcycle biomass unsuitable for human consumption, in this case from straw to beef, as suggested in van Hal et al. (2019). Thus, all emissions from wheat cultivation were allocated to the grain and the straw used as feed was considered 'free' from any environmental burden. The impact on the results of using this approach was tested in a sensitivity analysis by allocating emissions between the grain and straw based on the economic value of the two products. For straw, the value of straw used for bedding was assumed to be €0.1 kg^{-1} dry matter of straw (Jardstedt, 2019) assuming an exchange rate from Swedish krona (SEK) to € of 10:1. The economic value of wheat was set to €0.204 kg^{-1} of wheat grain (Swedish Board of Agriculture, n.d.a.), using the same exchange rate. The allocation factors in the sensitivity analysis resulted in 84 and 16%, for grain and straw respectively in the reference scenario, and 69 and 31% in the straw scenarios. For wheat cultivation on spared land in the straw-food scenario, the straw was assumed not to be used, but left in the field. For this wheat, all emissions were allocated to the grain.

Emissions of CO_2, CH_4, and N_2O were accounted for by weighting of these gases using the Global Warming Potential with a time horizon of 100 yr (GWP_{100}). Weighting factors were taken from the latest IPCC report (AR6) (fossil CH_4 29.8, biogenic CH_4 27.0, N_2O 273) (IPCC, 2021). Emission sources included were: CH_4 from enteric fermentation, CH_4 and N_2O from manure management, N_2O from soils (from feed production and grazing animals), CO_2 from soil carbon stock changes (from feed production), CO_2, CH_4, and N_2O from production of purchased products (feeds, synthetic fertilizers, breeding bulls, energy), and CO_2 from deforestation. More details on the climate impact calculations are given in the Supplementary materials (SM).

Methane emissions from enteric fermentation were calculated with a tier 3 approach (Swedish Environmental Protection Agency, 2021a), recommended for Swedish cattle by Bertilsson (2016) using methods from Nielsen et al. (2013) and Nielsen (2012). These methods are based on feed intake and nutritional values of the feed. Methane emissions from manure management were calculated with the tier 2 approach in IPCC (2019a) (Equation SM3), based on volatile solids content in excreted manure, which was also calculated according to IPCC (2019a). A more detailed description of these calculations can be found in the SM.

Direct and indirect emissions of N_2O from manure management were calculated with a tier 2 approach using emission factors from IPCC (2019a), based on nitrogen content in the manure, which was also calculated according to IPCC (2019a). Direct and indirect emissions of N_2O from manure on pasture were calculated based on emissions factors from IPCC (2019a). The fraction of nitrogen excreted on pastures was taken to be proportional to the time the cattle spent outdoors (i.e., all nitrogen excreted during the grazing period was assumed to be deposited on pasture). A more detailed description of these calculations can be found in the SM. Emissions of N_2O from land used for feed production were calculated using methods and emission factors from IPCC (2019b) (Table S3 in the SM), including both direct and indirect emissions. The amount of crop residues was calculated using biomass distribution data from Bertilsson and Nilsson (2020). Nitrogen content in above-ground crop residues from ley and whole crop silage was calculated based on their protein content. For the other crops, nitrogen contents in above-ground residues were based on the Swedish national inventory report (Swedish Environmental Protection Agency, 2021a). Nitrogen contents in below-ground residues were based on the same report (Swedish Environmental Protection Agency, 2021a).

Emissions from inputs (electricity, heating oil, diesel, synthetic fertilizers, purchased feed) were calculated using factors from various references (Table S4 in the SM). Values for energy demand for feed production were taken from Flysjö, Cederberg and Strid (2008).

Changes in soil carbon stocks as a result of the changes in crop production were modeled using the introductory carbon balance model (Andrén, Kätterer and Karlsson, 2004), which divides carbon into one old and one young carbon pool. The young pool can further be subdivided into three sub-pools receiving carbon from three different sources: above-ground crop residues (such as straw), below-ground residues (such as roots), and other carbon amendments (such as manure). Changes in total soil carbon stocks was calculated for a 30-yr period which is the time period the model was calibrated for (Andrén and Kätterer, 1997), with mean annual change in soil carbon over the 30 yr taken as yearly loss or sequestration of carbon, and hence of CO_2 to and from the atmosphere. Soil carbon stock changes were expressed as the difference between the studied scenario and the reference system, in which the soil was assumed to be in steady state (neither loosing nor sequestering carbon). Carbon stock change in soil under semi-natural pastures was not included, as it is relatively small (Karltun, Jacobson and Lennartsson, 2010). A more detailed description of the calculations of soil carbon stock changes can be found in the SM.

Emissions from potential deforestation to create more semi-natural pasture in the straw-food scenario were calculated based on average timber stocks for the Götaland region of 129 tons of dry matter per hectare (Swedish University of Agricultural Sciences, 2022). The tree biomass removed in deforestation was assumed to contain 50% carbon, which was emitted as CO_2. The emissions from deforestation were distributed over 100 yr, which was the same time period as was used for the GWP factors. Emissions from changes in soil carbon stocks on deforested soils were not included since these are uncertain and have been shown to be small in afforestation of grasslands in northern Europe (Bárcena et al., 2014).

Food production

The amount of meat produced per year was calculated based on annual number of animals slaughtered and their slaughter weight (Ahlgren et al., 2022). Meat production by-products in the form of offal and blood were accounted for using data from Strid, Wallin and Stenberg (2022). The amount of wheat grain produced was calculated based on yield and area. Energy, protein, and fat content in meat, offal, blood, and wheat (flour, bran, germ) were calculated based on data in the Swedish food database

(National Food Agency, 2023). The quantities of energy, protein, and fat obtained from animal sources (beef, offal, and blood) and plant sources (wheat) were considered separately, as they differ in terms of quality and digestibility (Bajželj, Laguzzi and Röös, 2021; Joye 2019). Food production was expressed as current use of offal, blood, and wheat as human food and also as maximum potential use as human food. At present, not all human-edible offal is used for human consumption (Strid, Wallin and Stenberg, 2022), nor is all wheat grain (Tillgren, 2021). Approximately 80% of wheat grain is consumed as refined wheat (Amcoff et al., 2012), which does not include the bran and the wheat germ. In addition, a substantial amount of wheat grain harvested in Sweden does not reach current quality standard for milling and is used as animal feed or for bioenergy. Data on the fraction of the harvested wheat that reaches milling standard are lacking, so a range of 65–100% was assumed, where 65% represented the current average situation and 100% a theoretical situation where quality standards are lowered so that all wheat can be used for human consumption, although not all in the form of high-rising white bread (with some intake as, e.g., muesli; Tillgren, 2021). In the maximum potential case, it was assumed that all wheat was consumed as wholegrain. For offal and blood, current utilization rates of blood and different offal from large slaughterhouses (Strid, Wallin and Stenberg, 2022) represented the current use, while for the maximum potential it was assumed that all blood and offal was used as food.

Results

Climate impacts

When carbon stock changes were excluded, the climate impact of feed production was 3.7, 3.4, and 3.2 kg CO_2e kg^{-1} carcass weight in the reference scenario, straw-food scenario, and straw-pasture scenario, respectively (Fig. 4). Thus, the straw-food scenario had 8% lower feed production impact than the reference scenario, while the straw-pasture scenario had 15% lower impact.

The climate impact of the whole farm (excluding carbon stock changes) was 210 tons CO_2e in the reference and straw-food scenarios, and 200 tons CO_2e in the straw-pasture scenario. Thus, the climate impact for the straw-food scenario was the same as that for the reference scenario, whereas the climate impact for the straw-pasture scenario was lower. On including emissions associated with deforestation and soil carbon stock changes in cropland the total climate impact was 220 and 210 ton CO_2e in the straw-food and straw-pasture scenarios, respectively (Fig. 5).

The climate impact of beef was 24 kg CO_2e kg^{-1} carcass weight in all three scenarios when deforestation and soil carbon stock changes in cropland were excluded. On including deforestation and soil carbon stock changes in feed production, the climate impact increased to 26 kg CO_2e in the straw-food scenario, while the climate impact in the straw-pasture scenario remained similar to that in the reference scenario (Fig. 5). Total carbon stock changes are included in the SM.

The climate impact of wheat was 0.27 kg CO_2e kg^{-1} dry matter in the reference scenario and 0.24 and 0.20 kg CO_2e kg^{-1} dry matter in the straw-food and straw-pasture scenarios, respectively, when soil carbon stock changes in feed production was excluded. On including carbon stock changes, the total impact decreased to 0.19 kg CO_2e kg^{-1} dry matter in the straw-food scenario and increased to 0.48 kg CO_2e kg^{-1} dry matter in the straw-pasture scenario.

Food production

The amount of food and macronutrients (beef, offal and blood, wheat) produced on-farm in the different scenarios are presented in Table 3. The reference and straw-pasture scenarios produced the same amount of food since the spared land was used for pasture in the straw-pasture scenario, with no additional food production. In the straw-food scenario, the spared land was used to grow more wheat, which led to a 79% increase in production of wheat and thereof more energy, protein, and fat.

Allocation of wheat emissions

When part of the emissions from wheat cultivation were allocated to the straw based on its economic value (using value for straw as bedding) instead of allocating all emissions to the grain, the

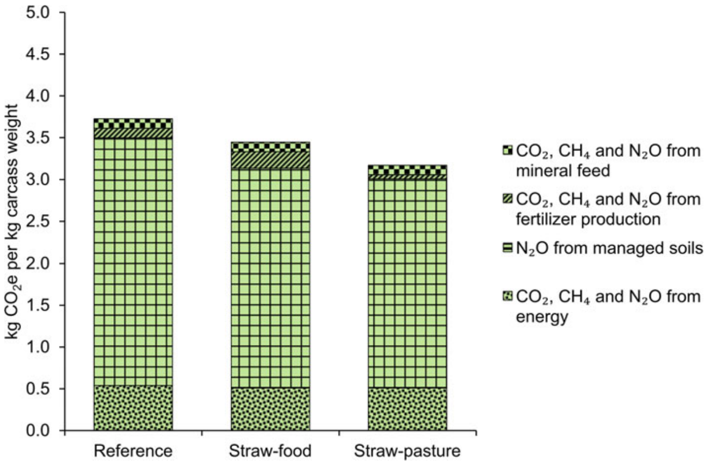

Figure 4. Climate impact of feed production (kg CO_2e kg^{-1} carcass weight beef) in the reference, straw-food, and straw-pasture scenarios, comprising N_2O emissions from managed soils and CO_2, CH_4, and N_2O emissions from mineral feed, fertilizer production, and energy use, but excluding emissions of CO_2 from soil carbon stock changes.

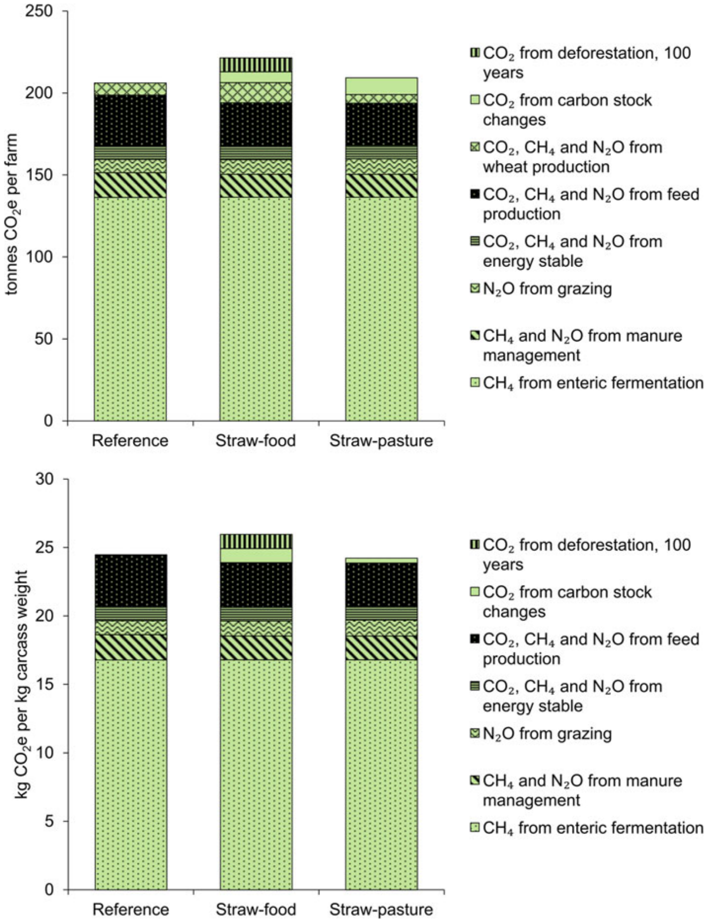

Figure 5. Climate impact of (top panel) the whole farm and (bottom panel) 1 kg of beef carcass weight in the reference, straw-food, and straw-pasture scenarios. The following emissions were included: CO_2 from deforestation (allocated over 100 yr), CO_2 from soil carbon stock changes, CO_2, CH_4, and N_2O from feed production and energy use in barns, N_2O from manure on pasture, CH_4 and N_2O from manure management and CH_4 from enteric fermentation.

Table 3. Amounts of food and macronutrients produced on-farm in the reference, straw-food, and straw-pasture scenarios

	Scenario		
	Reference	Straw-food	Straw-pasture
Beef (tons carcass weight)	11.6	11.6	11.6
Wheat whole grain (tons)	27	48	27
Nutrients from meat, offal, and blood			
Energy (GJ)	82.8–87.1	82.8–87.1	82.8–87.1
Protein (tons)	2.7–2.9	2.7–2.9	2.7–2.9
Fat (tons)	0.96–1.0	0.96–1.0	0.96–1.0
Nutrients from wheat			
Energy (GJ)	215–382	384–683	215–382
Protein (tons)	1.3–2.7	2.3–4.8	1.3–2.7
Fat (tons)	0.30–0.68	0.54–1.2	0.30–0.68

The higher ranges represent the maximum potentials with all wheat consumed as whole grain and the lower range represent the current average situation with 65% of the wheat kernel to human consumption.

Table 4. Climate impacts of beef and wheat production in the reference, straw-food, and straw-pasture scenarios when using different methods for allocation of emissions from wheat cultivation

	Impact from wheat all emissions allocated to grain	Impact from wheat emissions allocated between grain and straw	Difference (%)
Beef kg CO_2 kg^{-1} carcass weight			
Reference	24	24	0
Straw-food	26	26	+2
Straw-pasture	24	25	+2
Wheat kg CO_2 kg^{-1} dry matter			
Reference	0.27	0.23	−16
Straw-food	0.19	0.13	−42
Straw-pasture	0.48	0.32	−32

climate impact per kg carcass weight was scarcely affected, only increasing by a few percent in the two straw-scenarios (Table 4). The climate impact of wheat production was more strongly affected, with a reduction of 16, 46, and 32% in the reference, straw-food, and straw-pasture scenarios, respectively.

Discussion

Methane emissions from enteric fermentation in the rumen make up approximately two-thirds of the climate impact of beef from suckler herds. Methane emissions from enteric fermentation were similar in all scenarios despite changes in feed. There are several models for calculating methane emissions, possibly leading to varying results. The models used in this study are recommended for Swedish systems by Bertilsson (2016) and is used in the Swedish National Inventory Report (Swedish Environmental Protection Agency, 2021a). The model for suckler cows considers dry matter intake and concentration of fatty acids (Nielsen et al., 2013). If using another model for the suckler cows in this study, based on the model in Nielsen et al. (2013), which also takes into account the neutral detergent fiber concentrations in feed (which is higher in straw), emissions would be approximately 20% higher for all suckler cows. However, the differences between the scenarios would be only a few percent.

Feed production is also a major contributor to the climate impact of livestock products. When carbon stock and land use change were not included in the assessment, feed production in the reference scenario of this study had a lower climate impact compared with that reported by Moberg et al. (2019) for average Swedish meat from suckler herds. This was due to a lower use of mineral fertilizers in the present study.

Comparing the three scenarios in this study, the climate impact of feed production was lower in the straw-food and straw-pasture scenarios than in the reference scenario, as less ley needed to be harvested for winter feed when straw was used. Even though the emissions from feed production decreased by 8 and 15% in the straw-food and straw-pasture scenarios, feed-related impacts only made up approximately 15% of the total climate impact, thereof having only a small overall effect. The climate impact from feed production in straw-food decreased less than in straw-pasture since need for nitrogen was higher due to cropping of more wheat. Manure management emissions were slightly lower in the two straw-scenarios, since less manure excretion occurred indoors due to the reduced feed intake during indoor periods. However, the reductions from manure management indoors were offset by increased N_2O emissions from pasture, due to the greater amount of manure excreted on pasture. This, in turn, derived from higher feed intake on pasture. Nitrogen input from crop residues on new pasture also contributed to additional N_2O emissions.

When deforestation and soil carbon stock changes in cropland were considered in the analysis, the straw-food scenario had a greater total impact than the other scenarios. Grass-clover leys have a higher carbon input to soils than cereals (Börjesson et al., 2018). Consequently, growing more wheat at the expense of grass-clover ley, while also removing more straw from wheat cultivation (for use as feed), led to carbon losses from the soil in the straw-food scenario. This led to an increase in the climate impact of 7%. In the straw-pasture scenario, conversion of cropland to new pasture resulted in carbon emissions from soils due to lower yield from the new pasture compared with the previous grass-clover ley crop. This carbon loss from soil accounted for only a small proportion of total emissions from the farm in the straw-pasture scenario. This might seem counterintuitive, since conversion of cropland to pasture usually leads to carbon being sequestered in soil (Guo and Gifford, 2002). However, the land use change considered here was from high-yielding ley cultivation on cropland to more low-yielding pasture, resulting in a carbon loss from soil and hence net emissions of CO_2.

In the straw-food scenario, greater pasture area was needed due to higher feed intake on grazing compared with the reference scenario. As the spared cropland was used for wheat production in the straw-food scenario, new pasture had to be established and it was assumed that this would involve deforestation since most Swedish productive land not used as agricultural land is under forest (Statistics Sweden, 2019). However, in Sweden there are extensive areas of unused grassland or land currently used at low intensity that could be converted to pasture without causing emissions from deforestation. The emissions from deforestation are also influenced by the type of forest land converted, which affects factors such as dry matter content and land coverage that vary considerably (Swedish University of Agricultural Sciences, 2022). Moreover, semi-natural pastures provide multiple values (feed for cattle to produce beef, biological and cultural values) and how to allocate the impacts of deforestation between these values is open for debate. Normally, the impacts from restoration and maintenance of semi-natural pastures are allocated to the food product from the grazing livestock, considering only the food as valuable output. However, some studies have allocated part of the impacts to the cultural and biological values, with major effects on the results (Bragaglio et al., 2020; Kiefer, Menzel and Bahrs, 2015; von Greyerz et al., 2023). The time period over which deforestation emissions are allocated also impacts the results. When using a 100-yr allocation period as we did here, deforestation related emissions contributed with 4% to the total climate impact. If using, e.g., a 20-yr allocation period the contribution from deforestation would be five times higher, contributing with approximately 20% to the total climate impact. We did not include emissions from changes in soil carbon stocks due to deforestation as they were assumed to be small in this context (Bárcena et al., 2014). However, how soils are affected by different land uses is highly variable depending on a range of factors (e.g., soil type, climate, management). Such soil carbon changes could affect results in some situations.

This study evaluated two scenarios for the land ‘spared’ when using straw as feed: growing food (wheat) for humans or converting the cropland to pasture. When the land spared was used for wheat cultivation (straw-food scenario), on-farm production of wheat increased considerably, i.e., more food was produced on the same area. According to some previous studies, ‘number of people fed per hectare’ is an important sustainability indicator in a world with limited land resources and a growing population (Cassidy et al., 2013; Röös et al., 2021), so using the spared land to produce more food is a beneficial option from that perspective. Ultimately, however, agronomic and/or economic factors will determine whether growing wheat or other crops for direct human consumption can be considered feasible. In the forested regions of Sweden, fields are often small and scattered in the landscape, which can make cash cereal cropping challenging. In addition, farms in these regions are often small and do not have the necessary machinery, e.g., harvesters and drying facilities. A straw-food scenario may therefore not be suitable for all farms of the studied type. Another issue is that regional average wheat yields were assumed in this study, but farms with suckler herds are often located on marginal land, so wheat yield may have been overestimated. Predicting future conditions for different types of agricultural production is challenging due to, e.g., changing climate and uncertainty in markets due to the geopolitical situation.

When the land spared was converted to permanent pasture (straw-pasture scenario), the area of pasture on the farm increased and the potential climate impact of deforestation was avoided. However, converting cropland suitable for growing food into pasture would potentially increase food–feed competition.

How a farmer decides to use spared or available land will depend on many factors, such as economic incentives, personal preferences, and environmental conditions, which can change over time. Agri-environmental payment and support currently play a significant role in farmers’ income, and thus in their production choices. The payment and support can act as an economic incentive to create desirable farm characteristics, e.g., agri-environmental payments for maintaining semi-natural pasture can encourage grazing on these lands (Holmström et al., 2021). Other possible uses of spared land not considered in this study could be cultivating ley for biogas production (Gissén et al., 2014) to replace fossil energy on-farm, thereof potentially reducing emissions, or continued cultivation of ley for feeding an increased cattle stock, which would increase farm-level emissions considerably. All these factors vary over time, making it uncertain which scenario will prevail.

Moreover, inclusion of perennial leys in the crop rotation has many benefits, e.g., replacing some of the cereals in monoculture cereal production with perennial crops can improve soil health and reduce reliance on pesticides (Martin et al., 2020). The theoretical farm considered in this study was assumed to have over one-third of the crop rotation as grass-clover, which is a substantial proportion and well within what can be considered good practice (Karlsson et al., 2018).

To account for the value of upgrading straw to human-edible biomass, none of the climate impact from wheat cultivation was allocated to the straw in this study. To assess the effect of this allocation choice on the results, in sensitivity analysis 31% of the climate impact from wheat cultivation was allocated to the straw, and therefore to beef instead of wheat. This did not affect the climate impact of beef substantially (<2% increase in impact), since feed-related impacts only made up approximately 15% of the total climate impact of beef. However, the effect of allocation method on the climate impact will vary depending on proportion of straw in the total feed ration and the assumed economic value of the straw. In this study, the proportion of straw in total feed over the year was set to 11% for the suckler cows and was even lower for the entire system since only the suckler cows (and not the offspring) consumed straw. Allocating 31% of the climate impact to straw strongly affected the climate impact of wheat, however, which was reduced by up to 42% depending on scenario.

Data for feed intake in this study were collected from a few different data sources to cover all animal categories, but application on a farm in practice might look slightly different. For example, we included small amounts of pure grass silage in addition to grass-clover silage, but a farm might only have grass-clover leys and feed this forage harvested at different times to match the nutritional needs of different animal categories. However, we believe that this would not have major influence on the results.

A challenge with straw feeding is to ensure that it works well in practice. The average suckler herd in Sweden has only 21 cows and 90% of all farms with suckler herds have fewer than 50 suckler cows (Swedish Board of Agriculture, 2022). Feeding straw to small herds may involve alternating between providing whole bales of either straw or grass-clover silage on a daily basis. However, this method has been shown to be unsuitable for some animals, e.g., some cows might consume substantial amounts on days they are fed silage, which can result in digestive problems (Dahlström and Arnesson, 2016). Another way to feed straw is by mixing straw and grass-clover silage in a mixer or full feed wagon. The mixing makes sorting out the preferred grass and clover particles more difficult for the animals, resulting in higher and steadier intake of straw (Dahlström and Arnesson, 2016). However, farmers with herds of fewer than 100 suckler cows do usually not have the financial resources to invest in a mixer/full feed trailer. The potential to scale up straw feeding in Sweden thus depends on availability, price, and practical scope for good implementation.

Upgrading straw to meat using beef cattle did not yield any notable climate benefit. However, there are other potential positive effects. Using agricultural residues as feed and thereof upgrading it to high-value food increases its value (Mottet et al., 2017). Replacing parts of the grass-clover silage with straw can consequently be a way to refine these materials into valuable food products, adding value to these agricultural residues. This approach reduces the need to produce additional feed, thereby decreasing cropland use while maintaining the same level of food production. The spared land can be repurposed, potentially allowing more food to be grown and feeding more people on the same area of land. Additionally, this feeding strategy increased the use of semi-natural pastures using the same number of cattle, showing potential to maintain more semi-natural pastures (important for biodiversity conservation in Sweden; Eriksson 2022) without increasing the number of animals (hence keeping methane emissions down). However, removing straw from soils resulted in CO_2 emissions due to losses of soil organic carbon. Nonetheless, Björnsson and Prade (2021) indicate that measures can be implemented to prevent losses in soil organic carbon when removing cereal straw.

Conclusions

On the theoretical Swedish farm considered in this study, using a combination of cereal straw and grass-clover silage as winter feed for suckler cows reduced the climate impact associated with feed production compared with feeding only grass-clover silage. However, it also reduced feed intake indoors and increased biomass intake on pasture during the grazing season, and hence increased the grazing area required. The total climate impact of beef per kg of carcass weight, and of the farm as a whole, increased when the additional grazing area required generated by deforestation. If deforestation was not necessary, e.g., when grazing was established on the cropland spared by replacing some grass-clover silage with straw (straw-pasture scenario), the climate impact was comparable to that of beef from suckler cows exclusively fed a grass-clover mixture during the winter (reference scenario). Therefore, upcycling the by-product straw to meat had no notable effect on the climate impact, indicating that using crop residues as feed does not always entail a climate benefit. However, increasing the area of pasture can have a positive effect on biodiversity if more biologically rich semi-natural pastures are maintained or restored. When the land spared by replacing some grass-clover silage with straw as winter feed was used for increased wheat production (straw-food scenario), total food production from the system increased, which can have indirect positive benefits, e.g., potentially spared emissions from food production elsewhere.

Supplementary material. The supplementary material for this article can be found at https://doi.org/10.1017/S1742170524000255.

Acknowledgments. We would like to thank Kristina Holmström at Hushållningssällskapet Sjuhärad/the Department of Animal Environment and Health, Swedish University of Agricultural Sciences, for calculating the suckler cows' intake of the mixture of grass-clover and straw.

Author contributions. K. G.: conceptualization, methodology, formal analysis, writing—original draft, visualization. N. E.: conceptualization, methodology, writing—review and editing, supervision. M. J., A. H., and K. A. S.: conceptualization, resources, writing—review and editing. E. R.: conceptualization, methodology, writing—review and editing, supervision, project administration, funding acquisition.

Funding statement. This work was supported by Mistra (K. G., N. E., M. J., A. H., K. A. S., and E. R., Mistra Food Futures DIA 2018/24 #8); and Interreg Öresund-Kattegatt-Skagerrak European Regional Development Fund (A. H. and M. J. Grant number ÖKS-4-2011-47); and Region Västra Götaland, Agroväst (A. H. and M. J.); and the Swedish University of Agricultural Sciences (A. H. and M. J.).

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Acta Universitatis Agriculturae Sueciae

Doctoral Thesis No. 2026:28

Cattle systems have a high climate impact and require mitigation, but they are also multifunctional, providing benefits beyond food that may be overlooked in mitigation efforts. This thesis increases knowledge about how climate impacts from beef and milk production can be quantified and reduced whilst considering system multifunctionality and exploring trade-offs. It shows that assessing mitigation in a broader context requires a systems perspective that integrates emissions, ecosystem services, and food production.

Karin von Greyerz received her graduate education at the Department of Energy and Technology, Swedish University of Agricultural Sciences (SLU), Uppsala, and holds a Master of Science in Environmental and Water Engineering from Uppsala University.

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ISSN 1652-6880
ISBN (print version) 978-91-8124-245-4
ISBN (electronic version) 978-91-8124-275-1