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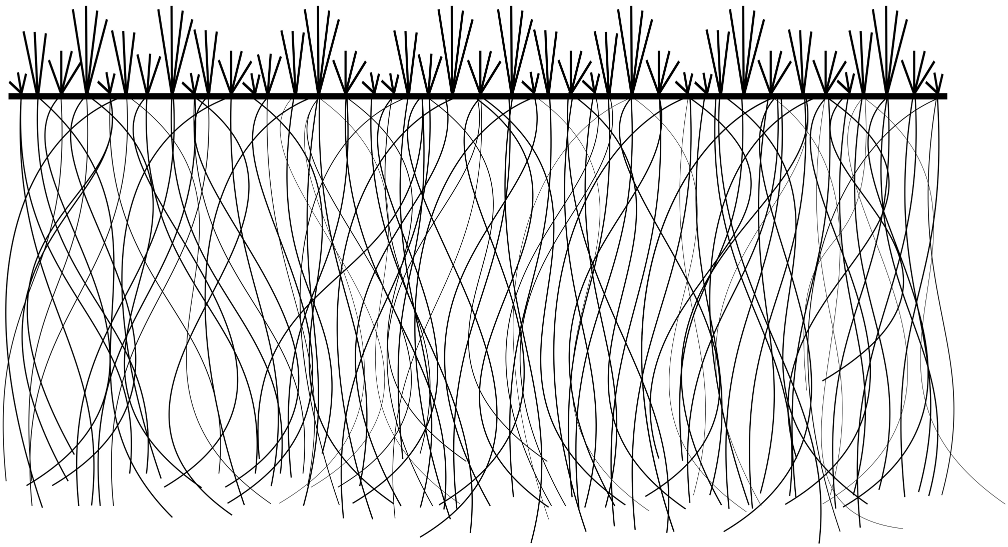
A life cycle perspective

Johan Nilsson



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Abstract

One strategy to mitigate climate change and increase energy security is to replace fossil fuels with bio-based alternatives. Upgraded biogas (biomethane), is a renewable energy carrier that can be readily integrated into existing infrastructure, *e.g*. for heat and electricity generation, or as vehicle fuel. Life Cycle Assessment (LCA) is a frequently used methodology for studying the environmental impact of bioenergy systems, but spatial and temporal variations in emissions, emissions from soil organic carbon dynamics and nitrous oxide emissions are often inadequately described in LCA.

In this thesis, LCA methodology was used to explore the climate mitigation potential of grass and cover crop cultivation and their integration into biogas systems in Sweden. Various approaches were employed in data inventory, including agro-ecosystem modelling, simple carbon modelling, empirical approaches for estimating nitrous oxide emissions, and data from short- and long-term field experiments. Alongside the conventional GWP metric, a dynamic impact assessment method was applied to consider the timing of emissions.

The results revealed considerable mitigation potential for grass- and cover crop based-biogas systems. Introducing a grass-based biogas system using fallow land in Uppsala Municipality doubled the biogas production, leading to mitigation potential of 9950 tonnes CO_2-eq per year. However, the biogas mitigation potential exhibited large variation (79 to 102% compared with diesel fuel), depending on where in the region grass was cultivated. Cover crop cultivation had higher climate mitigation potential when the cover crop was harvested, primarily through fossil fuel substitution and a reduced risk of elevated nitrous oxide emissions during winter. These findings offer valuable insights that can hopefully be used in creating sustainable crop-based bioenergy systems in Sweden and other regions with similar conditions.

Keywords: biomethane, DNDC model, energy crops, greenhouse gas emissions, ICBM, intermediate crop, ley cultivation, soil N_2O emissions, soil organic carbon

Odling av vall och mellangrödor för ökad biogasproduktion och minskade utsläpp av växthusgaser

Sammanfattning

En viktig strategi för att begränsa den globala uppvärmningen och öka energisäkerheten är att ersätta fossila bränslen med biobaserade alternativ. Uppgraderad biogas (biometan) är en förnybar energibärare som lätt kan integreras i befintlig infrastruktur, t.ex. för värme- och elproduktion eller som fordonsbränsle. Livscykelanalys (LCA) är en lämplig metod att använda för att studera miljöpåverkan av en produkt eller tjänst. I LCA-metoden tas däremot sällan tids- och platsberoendet med i bedömningen. Dessutom beskrivs ofta markprocesser som förändring av markens kollager och utsläpp av lustgasemissioner bristfälligt.

I denna avhandling användes LCA-metodik för att undersöka potentialen för att minska klimatpåverkan genom odling av vall och mellangrödor och deras integrering i biogassystem i Sverige. Olika metoder användes för datainventering, inklusive simulering med hjälp av statistiska och process-baserade modeller samt data från fältexperiment. Vid sidan av den konventionella GWP-metoden användes en dynamisk metod för att bedöma klimatpåverkan för att ta hänsyn till tidpunkten för utsläppen.

Resultaten visade på en betydande utsläppsminskning för biogassystem jämfört med fossila bränslen. Införandet av ett vallbaserat biogassystem i Uppsala kommun fördubblade biogasproduktionen och resulterade i en utsläppsminskning motsvarande 9950 ton CO_2-eq per år. Biogasens utsläppsminskning uppvisade dock stora variationer (79 till 102 % jämfört med diesel), beroende på var i regionen vallen odlades. Odling av mellangrödor hade högre potential att minska klimatpåverkan när mellangrödan skördades, främst genom ersättning av fossila bränslen och en minskad risk för förhöjda lustgasutsläpp under vintern. Dessa resultat ger värdefulla insikter som förhoppningsvis kan användas för att skapa hållbara bioenergisystem i Sverige och andra regioner med liknande förhållanden.

Keywords: biometan, DNDC, energigrödor, växthusgasutsläpp, ICBM, mellangröda, vallodling, lustgasemissions, markkol

Dedication

To my family and friends.

"Prediction is very difficult, especially if it's about the future"

- Niels Bohr

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. Nilsson, J., Tidåker, P., Sundberg, C., Henryson, K., Grant, B., Smith, W. & Hansson, P-A. (2020). Assessing the climate and eutrophication impacts of grass cultivation at five sites in Sweden. *Acta Agriculturae Scandinavica, Section B — Soil & Plant Science* 70 (8), 605-619.

II. Nilsson, J., Sundberg, C., Tidåker, P. & Hansson, P-A. (2020). Regional variation in climate impact of grass-based biogas production: A Swedish case study. *Journal of Cleaner Production* 275, 122778.

III. Nilsson, J., El Khosht F. F., Bergkvist, G., Öborn I. & Tidåker, P. (2023). Effect of short-term perennial leys on life cycle environmental performance of cropping systems: An assessment based on data from a long-term field experiment. *European Journal of Agronomy* 149, 126888

IV. Nilsson, J., Ernfors, M., Prade, T. & Hansson, P-A. (2023). Cover crop cultivation strategies for climate change mitigation and biogas production (under review)

Papers I-III are reproduced with the permission of the publishers (open source).

The contribution of Johan Nilsson to the papers included in this thesis was as follows:

I. Planned the study and developed the modelling approaches together with the co-authors, performed the modelling and analysed the data. Prepared the data presentation, including figures and tables, and wrote the manuscript with support from the co-authors.

II. Planned the study in collaboration with the co-authors. Carried out the modelling and data analysis. Prepared the data presentation, including figures and tables, and wrote the manuscript with support from the co-authors.

III. Performed the LCA and prepared the data presentation, including figures and tables. Wrote the manuscript with support from the co-authors.

IV. Planned the study together with the co-authors. Performed the LCA and prepared the data presentation, including figures and tables. Wrote the manuscript with support from the co-authors.

Papers produced but not included in this thesis:

V. **Nilsson, J.** & Martin, M. (2022). Exploratory environmental assessment of large-scale cultivation of seaweed used to reduce enteric methane emissions. *Sustainable Production and Consumption* 30, 413-423.

VI. Brandão, M., Ekvall, T., Poulikidou, S., Johansson, K., **Nilsson, J.**, Nojpanya, P., Wikström, A. & Rydberg, T. (2022). RED, PEF, and EPD: Conflicting rules for determining the carbon footprint of biofuels give unclear signals to fuel producers and customers. *Frontiers in Climate* 4, 988769

VII. Ekvall, T., Gottfridsson, M., Nellström, M., **Nilsson, J.**, Rydberg, M. & Rydberg, T. (2021). Modelling incineration for more accurate comparisons to recycling in PEF and LCA. *Waste Management* 136, 153-161.

Abbreviations and clarifications

AGTP	Absolute global temperature potential
AGWP	Absolute global warming potential
Cover crops	Crops grown between main crops in a crop rotations
CU	Cereal units
CDR	Carbon dioxide removal
DM	Dry matter
DNDC	DeNitrification DeComposition model
EPD	Environmental product declaration
ER	Energy ratio
GHG	Greenhouse gas
Grass	Perennial grasses and legumes cultivated in either mixed or pure stands
GTP	Global temperature potential
GWP	Global warming potential
ha	Hectare (10^4 m^2)
ICBM	Introductory Carbon Balance Model
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
LCI	Life cycle inventory

J	Joule
PEF	Product environmental footprint
SOC	Soil organic carbon

1. Introduction

We are currently at the onset of a human-induced climate crisis. The Earth's average temperature has increased by approximately 1.1 °C since the end of the 19th century (IPCC, 2023). This global warming is already causing severe damage and loss to nature and humans. Rising sea levels, more frequent and severe heat waves, droughts and storms are resulting in widespread crop failure, biodiversity loss and increased conflict over resources. The projected further increase in temperature is expected to worsen this situation, triggering more adverse and irreversible effects. To avoid this, immediate curbing of greenhouse gas (GHG) emissions is needed in all sectors, requiring concerted efforts from governments, businesses and individuals around the world (IPCC, 2023).

One strategy to mitigate climate change is to phase out fossil fuels (such as oil, natural gas and coal) and replace them with bio-based alternatives, in a process commonly referred to as transition from a fossil economy to a bioeconomy[1]. In Sweden, one of the greatest challenges in this transition lies in the transport sector, where the majority of all fuels used are fossil-based (SEA, 2022). Beyond reducing the climate impact, replacement of fossil fuels can enhance domestic energy security and reduce exposure to price volatility, as fossil resources are highly susceptible to geopolitical tensions. This was recently illustrated by the full Russian invasion of Ukraine in 2022, which led to a spike in global energy prices (Mišík, 2022).

Biogas is a competitive biofuel primarily obtained from anaerobic digestion of organic materials. In addition to producing energy, the anaerobic digestion process produces digestate, which can be utilised as organic

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[1]The term "bioeconomy" has been used for several decades, but gained prominence in the late 20th and early 21st centuries as discussions on sustainable development and the use of biological resources increased. Today it is widely employed across diverse sectors to describe activities in sustainable utilisation of biological resources.

fertiliser, replacing synthetic fertilisers while adding carbon to the soil. Several studies have suggested that grass crops are well-suited as feedstock for biogas systems (Smyth *et al*., 2009; Börjesson & Tufvesson 2011; Auburger *et al*., 2017). Here, I refer to grass as perennial grasses and legumes cultivated either in mixed or pure stands. Perennial crops, such as grass, also have a greater capacity to sequester carbon in the soil compared with annual crops (Bolinder *et al*., 2010). Soil organic carbon (SOC) sequestration, *i.e.* the capture and storage of carbon dioxide (CO_2) from the atmosphere in soils as organic carbon, has been advocated as a cost-efficient strategy with high potential to mitigate climate impacts (Minasny *et al*., 2017).

Cultivation of feedstock for bioenergy production requires agricultural land, which is a limited resource. However, agricultural activities can be expanded by tapping unused potential within agricultural systems, such as set-aside land not currently used for farming purposes (Tilman *et al*., 2009; Carlsson et al. 2017; Prade et al. 2017). Another approach for cultivation of energy crops with low competition with food production is to grow these crops between the main crops in crop rotations, where they are often referred to as cover crops, intermediate crops or catch crops (Aronsson *et al*., 2023). In this thesis, the term cover crop is used to refer to this particular type of cultivation. Similarly to grass cultivation, growing cover crops is associated with SOC sequestration (Poeplau & Don, 2015; Abdalla *et al*., 2019). In addition, grass and cover crops can be used to promote ecosystem services and thereby enhance resilience by fostering yield stability and reducing reliance on agronomic inputs, such as chemical fertilisers and herbicides (Torstensson & Aronsson, 2000; Blanco-Canqui *et al*., 2015; Bowles *et al*., 2020; Tamburini *et al*., 2020; MacLaren *et al*., 2022). Reducing reliance on agronomic inputs could increase cost-efficiency, reduce environmental impacts (Tidåker *et al*., 2016) and bolster regional security in terms of the supply of agricultural goods. However, further knowledge is needed to fully grasp the bioenergy potential, its impact on the agricultural system, and the environmental consequences resulting from cultivating unused and underused agricultural land across different geographies and scales.

When assessing the environmental impact of crop-based bioenergy systems, it is important to apply a life cycle perspective (Creutzig *et al*., 2015). Life cycle assessment (LCA) has become a widely accepted method for evaluating the environmental implications of agricultural and energy systems, gaining recognition from policymakers in both public and private organisations (Brandão *et al*., 2022b). In LCA, emissions and resources used throughout the whole (cradle-to-grave) or parts (*e.g.* cradle-to-gate) of the life cycle of a product or process are considered (ISO, 2006a, 2006b). Assessing crop-based systems is often complex because of the inherent spatial and temporal variability of agriculture, including weather conditions, soil properties, crop rotations and transport distances. Consequently, the environmental impact can differ substantially depending on where and when cultivation occurs (Humpenöder *et al*., 2013; Hörtenhuber *et al*., 2014; Hammar *et al*., 2017; Henryson *et al*., 2019). Unfortunately, conventional LCA studies frequently overlook the influence of variation in cultivation properties between sites and over time, leading to large uncertainties that can undermine the credibility and utility of the LCA results (Notarnicola *et al*., 2017; Patouillard *et al*., 2018). To enhance the reliability of agricultural LCAs, incorporation of spatial and temporal differentiation in emissions modelling and impact characterisation has been recommended (Reap *et al*., 2008; Notarnicola *et al*., 2017), as has considering crop interactions within the cropping system (Goglio *et al*., 2018a).

2. Aim and structure

2.1 Aim

The overall aim of this thesis was to explore the climate change mitigation potential of grass and cover crop cultivation and its integration into biogas systems in Sweden. This was done using LCA methodology in combination with agro-ecosystem modelling and data from field experiments. Specific objectives were to analyse the:

i. Temporal life cycle climate impact of grass cultivation at different fertiliser intensities and under different cultivation conditions due to spatial variation (Paper I & II).
ii. Climate effect of integrating grass and cover crops in cropping systems through crop rotation diversification and cover cropping (Paper III & IV).
iii. Life cycle impact of biogas production systems using biomass from grass cultivation distributed in a landscape (Paper II) and oilseed radish as a cover crop (Paper IV).

2.2 Work and thesis structure

This thesis is based on the work presented in Papers I-IV, which collectively explore the effect on the environment of exploiting the potential for biomass production in Swedish agriculture to facilitate biogas production. The research focused on different forms of potential for biomass production, where Papers I and II investigated grass cultivation and grass cultivation on unused agricultural land, Paper III investigated inclusion of grass in crop rotations, and Paper IV investigated the cultivation of oilseed radish as a cover crop (*Figure 1*).

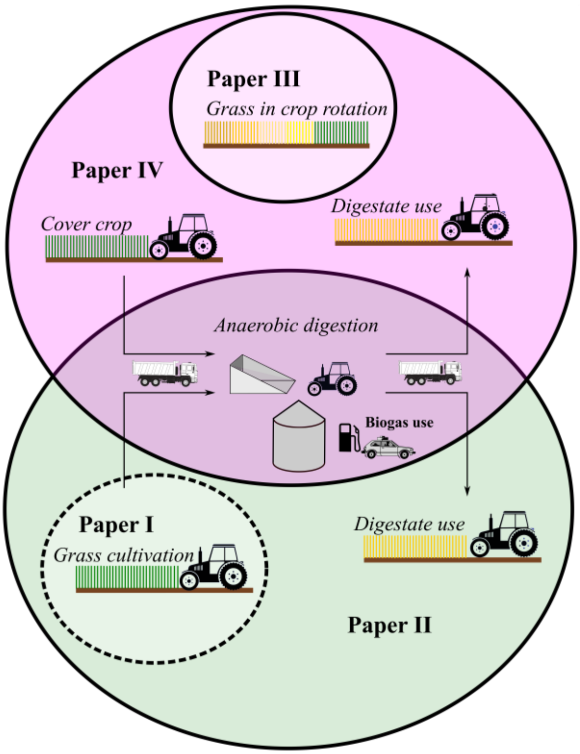


Figure 1. Schematic illustration of the work reported in Papers I-IV and the links between the papers.

A thematic overview of the work performed is presented in (Table 1). In Paper I, the climate and eutrophication impact of grass cultivation were investigated at five sites in southern and central Sweden. The study focused on the gross environmental effect of grass cultivation (cradle to farm-gate), without an applied reference scenario. In Paper II, the analysis was expanded to include handling of the grass biomass produced for biogas production and use of the residual digestate as fertiliser (cradle to grave) and was performed on regional level (Uppsala Municipality), including over 1000 spatially distributed sites with individual soil properties delivering biomass to a central biogas plant. The net climate effect of the biogas system was assessed by including a reference system, which was replaced by the grass-based biogas system. In Paper III, the environmental impacts (climate, eutrophication and energy resource depletion) of including grass in crop rotations were investigated. The study was performed using data from a long-term field experiment in Sweden where three different six-year crop rotations have been applied since the 1960s: (i) two-year grass-legume mixture, (ii) two-year pure grass and (iii) without grass. The study in Paper IV assessed the climate effect of introducing cover crops (oilseed radish) into cropping systems. In the assessment, a reference scenario (without cover crop) was compared with three different management strategies: (i) leaving the cover crop over winter and ploughing it under in spring, (ii) harvesting the aboveground biomass during autumn, and (iii) harvesting both the aboveground and belowground biomass during autumn. In the alternative scenarios (ii) and (iii), the harvested biomass was assumed to be transported to a biogas plant, where it was converted to upgraded biogas used to replace fossil diesel fuel in the transport sector.

Table 1. *Overview of the papers included in this thesis.*

	Paper I	Paper II	Paper III	Paper IV
Specific aim addressed	1	1, 3	2	2, 3
Case study	Grass cultivation at five sites in Sweden	Grass-based biogas using fallow land in Uppsala Municipality	Grass cultivation (pure grasses and grass-legume mixtures) in crop rotation	Different strategies for cover crop cultivation, including biogas production
Impacts	Climate; eutrophication	Climate; energy	Climate; eutrophication; energy	Climate
Mitigation potential	SOC sequestration	SOC sequestration; energy substitution	Resource efficiency; SOC sequestration	Resource efficiency; SOC sequestration; energy substitution
LCI for SOC changes and soil N_2O emissions from:	DNDC	DNDC	Long-term field experiment; IPCC Tier I	ICBM, IPCC Tier I with crop and digestate specific EF

3. Background

3.1 Climate change and climate change mitigation

The greenhouse effect refers to the phenomenon where certain gases, known as greenhouse gases (GHGs), trap infrared thermal radiation within the atmosphere, resulting in warming of the Earth's surface. The current accumulation of GHGs in the atmosphere is leading to increased retention of outgoing thermal radiation, causing a distortion in the global energy balance. This distortion is known as radiative forcing and is expressed in W m^{-2}. The radiative forcing amplifies the greenhouse effect, resulting in the entrapment of more energy within the atmosphere, ultimately leading to a rise in average global temperature and subsequent climate change (Myhre *et al*., 2013). The international community has adopted various climate change mitigation targets to limit the extent of global temperature rise. The most significant of these is the Paris Agreement, which was signed by member states of the United Nations Framework Convention on Climate Change (UNFCCC) during the 21st Conference of the Parties (COP 21). The Paris Agreement states that global warming must be restricted to well below 2 °C by the end of the century, with efforts made to stay below 1.5 °C (UNFCCC, 2016).

In 2017, the Swedish Parliament implemented a climate policy framework aimed at achieving GHG emissions cuts of 63% by 2030 and 75% by 2040 relative to 1990, levels and no net territorial GHG emissions by 2045. The corresponding target deadline for no net emissions for the European Union (EU) is 2050 (EU, 2021). The Swedish commission tasked with evaluating progress towards the targets has repeatedly reported that the stated targets are not achievable under current policies (Climate Policy Council, 2023). Similar concerns have been raised regarding the Paris

Agreement targets (Rogelj *et al*., 2016; Peters *et al*., 2017). Moreover, in 2023, the World Meteorological Organization (WMO) indicated that the 1.5 °C target is likely be transgressed for the first time during this decade (WMO, 2023). To effectively achieve the set targets, immediate and substantial reductions in emissions across all sectors, particularly those originating from fossil sources, are imperative.

The agricultural sector is estimated to account for 22% of all anthropogenic GHG emissions (IPCC, 2022), and is a key driver of increasing pressure on several other planetary boundaries (Foley *et al*., 2011; Steffen *et al*., 2015; Campbell *et al*., 2017). At the same time, the agricultural sector is one of the most sensitive sectors when it comes to environmental impacts, such as climate change and biodiversity loss (Raven & Wagner, 2021; Kornhuber *et al*., 2023). The Intergovernmental Panel on Climate Change (IPCC) essentially mentions three categories of mitigation potential in the agricultural sector (IPCC, 2019b). The first (1) group of measures involve transitioning consumption patterns towards agricultural products associated with low GHG emissions. This can be done by *e.g.* promoting more plant-based foods in regions with diets consisting of a high proportion of meat and dairy, as animal-based foods generally generate higher GHG emissions than plant-based foods (Bajželj *et al*., 2014; Röös *et al*., 2020). The second (2) group of measures involve reducing waste flows in the agricultural sector. Considerable wastage in the sector is leading to substantial loss of resources and energy in wasted agricultural production, resulting in elevated impact per product that actually comes into use (Bajželj *et al*., 2014; Zhu *et al*., 2023). The third (3) group of measures, which are the primary focus in this thesis, pertain to adoption and adaptation of agricultural practices aimed at reducing the climate impact of agricultural production systems. This group may be further divided into three subgroups, all of which are included in this thesis. The first subgroup a) entails measures aimed at reducing direct emissions in agriculture, *e.g.* through reduced use of fossil fuels, increased resource efficiency and reduced soil-borne nitrous oxide (N_2O) emissions. The second subgroup b) entails measures aimed at sequestering carbon in vegetation and soils, creating CO_2 sinks within the system. Lastly, the third subgroup c) involves providing biomass to replace GHG-intensive products, such as fossil fuels or materials (IPCC, 2019b).

3.2 Grass and cover crops to reduce reliance on input commodities in agricultural systems

Since the *Green Revolution*[2], synthetic fertilisers and pesticides have been used to increase food production for an increasing human population (MacLaren *et al*., 2022). The introduction of these inputs into agricultural practices allowed farmers to specialise in a few crops and abandon the diverse crop rotations that had characterised European agriculture since the introduction of perennial grass-clover rotations in the 19th century. However, the introduction of these inputs also increased the environmental impacts of the sector (Campbell *et al*., 2017; Tang *et al*., 2021). Furthermore, the heavy reliance on these inputs makes the agricultural system vulnerable to instability in the geopolitical landscape, which may call for the re-introduction of more diversified agriculture.

One strategy to reduce the dependence on agricultural inputs is to promote ecosystem services by increasing the diversity in cropping systems (Nemecek *et al*., 2015; Tamburini *et al*., 2020). Earlier studies have shown that increased crop diversification can keep crops healthier, increase nutrient delivery, increase crop yield and reduce yield losses due to weather extremes (Gaudin *et al*., 2015; Bowles *et al*., 2020).

Including perennial crops, such as temporary grasses, in cereal-dominated cropping systems has been demonstrated *e.g.* to reduce dedicated pests of annual crops (Kirkegaard *et al*., 2008). In addition, integration of leguminous crops together with grasses in cropping systems has the ability to provide substantial amounts of nitrogen to the system via biological fixation of nitrogen from the atmosphere through the symbiotic association between legumes and nitrogen-fixing bacteria (Carlsson & Huss-Danell, 2003; Peoples *et al*., 2019). Finding a sustainable nitrogen source is a key challenge to achieving a sustainable agricultural sector. By including legumes in cropping systems, the dependence on chemical nitrogen fertilisers (a highly resource-intensive agricultural input) can be effectively reduced (Ledgard & Steele, 1992). It is widely recognised that nitrogen fertilisers have a significant environmental impact (IEA, 2021). Therefore, inclusion of

[2]The *Green Revolution* refers to a period of agricultural modernisation that began in the mid-20th century and was aimed at increasing global food production. This movement was characterised by the introduction of high-yielding crop varieties, chemical fertilisers and pesticides, which dramatically increased yields in many parts of the world.

legumes via grass cultivation serves as a strategic approach for addressing this challenge and advancing the sustainability of agricultural systems.

Grass is cultivated in either permanent stands or temporary leys. In temporary leys, the grass undergoes regular re-sowing in crop rotations to maintain its productivity (Allen *et al*., 2011). Traditionally, grass is primarily used as fodder, but there have been ongoing discussions about alternative applications, such as protein extraction or utilisation as feedstock for biofuel production (Tilman *et al*., 2006; Auburger *et al*., 2017; Carlsson *et al*., 2017; Santamaría-Fernández *et al*., 2017). Grass is typically grown as a mixture of different species, occasionally including clover or other leguminous species. Incorporating multiple species offers advantages, as they can effectively occupy diverse niches, both spatially and temporally (Tilman *et al*., 1997). Consequently, well-designed species mixtures often produce higher biomass yields than monocultures (Tilman *et al*., 2006; Picasso *et al*., 2011).

Another strategy to increase diversification in cropping systems is to add a crop between the main crops in a crop rotation. In this thesis, the term cover crop is used to describe this cultivation method. Similarly to grass cultivation, cover crops can bring several benefits to the cropping system, such as reduced nutrient leaching, improved soil structure and increased (SOC) stock (Torstensson & Aronsson, 2000). An ideal cover crop should be easily established and exhibit vigorous autumn growth to capture and retain nutrients, preventing leaching. It should also possess the ability to improve soil structure and compete with weeds (Aronsson *et al*., 2012). Several species, such as brassicas, have the potential to serve as cover crops. Brassicas are typically sown immediately after the main crop harvest. One advantage of using brassicas in cover cropping is their rapid growth rate, which effectively reduces nutrient leaching from the soil, especially under favourable conditions (Aronsson *et al*., 2012). Certain brassica crops produce a rapidly spreading taproot that delves deep into the soil. This characteristic enables cover crops of this type to extract nutrients from lower soil layers, resulting in further reductions in nutrient leaching. Moreover, the taproot system of brassicas contributes to soil loosening and enhances overall soil structure (Blanco-Canqui *et al*., 2015). Oilseed radish, belonging to the *Brassica* genus, is commonly cultivated as a cover crop. While most oilseed radish varieties demonstrate tolerance to clubroot, they are not entirely resistant. Therefore, to minimise risks, it is advised to avoid cultivating oilseed radish as a catch crop in close proximity to other crops susceptible to clubroot within the crop rotation (Aronsson *et al*., 2012).

3.3 Use of agricultural soils as atmospheric carbon sinks

Since the beginning of agriculture, soils have been a source of atmospheric CO_2 due to depletion of stored carbon (Lal, 2010). The rate of depletion has been accelerated by the specialisation of arable agriculture, with systems dominated by annual crops (Bolinder *et al*., 2010). By implementing agricultural management practices that promote carbon sequestration, it is possible to partially counteract the carbon loss and establish agricultural soils as carbon sinks (Rumpel *et al*., 2020). Previous studies have shown that SOC is more abundant in perennial cropping systems than in annual systems. This has been attributed to greater root production, less exposure to ploughing and longer growing season (Baker *et al*., 2007; Bolinder *et al*., 2010; Kätterer *et al*., 2011; Börjesson *et al*., 2018). Similar results have been found for cultivation of cover crops (Poeplau & Don, 2015; Abdalla *et al*., 2019), where the photosynthesis during the vegetation period is more efficiently utilised by cultivating an additional crop after harvest of the main crop in the rotation. This increases the input of carbon to the soil and thus increases potential SOC sequestration.

Most suggested pathways for meeting the current climate mitigation targets comprise carbon dioxide removal (CDR) (Babiker *et al*., 2022), which includes technologies, practices and approaches that remove and isolate GHGs from the atmosphere with the intention of mitigating climate change (Minx *et al*., 2018).

Soil organic carbon sequestration through changes in agricultural practices is one CDR that is attracting growing interest (Minx *et al*., 2018). One example is the “4per1000” initiative, which was launched at COP 21 with the objective of promoting SOC sequestration as an important tool in climate mitigation schemes. The name of the initiative originates from the calculation that if SOC storage were to increase by 0.4% per year, human-induced CO_2 emissions at today’s levels would be offset (Minasny *et al*., 2017). Increasing SOC storage is not only beneficial for climate change mitigation, but also improves soil quality, *e.g.* through increased water-holding capacity, a more steady supply of nutrients, improved soil structure and reduced risk of soil compaction (Lal, 2004).

Soil organic carbon storage is a balance between carbon inputs, in the form of roots, crop residues *etc.*, and carbon outputs, in the form of organic matter degradation and carbon leaching. For soils that are theoretically in equilibrium, *i.e.* where carbon inputs are equal to carbon outputs, an increase in carbon inputs will result in an increase in SOC stock and carbon sequestration. The carbon stock will continue to increase until the soil reaches a new dynamic equilibrium, which can take a long time (Smith *et al*., 2008), especially in the cold climate in Sweden (Kätterer *et al*., 2012). The carbon stock level at which the soil reaches the new equilibrium depends on spatially differentiated properties such as soil characteristics, climate, type of crop and management. This means that SOC sequestration will always have a finite climate mitigation capacity (Smith, 2014), and that the effect will vary between different locations and between different points in time for a particular mitigation scheme (Kätterer *et al*., 2012). Furthermore, SOC sequestration is a reversible process, which means that sequestered carbon can be re-emitted to the atmosphere at any time, *e.g.* if the continuity in land management is broken. Soil carbon loss typically happens faster than soil carbon build-up (Smith, 2005).

3.4 Biogas production and fossil fuel substitution

Biogas, a well-established biofuel with historical roots dating back to at least the mid-nineteenth century, is generated through microbial decomposition of organic material under anaerobic conditions (Bond & Templeton, 2011). The composition of the gas resulting from this process varies depending on the input materials, but typically consists of 50-70% methane (CH_4), 25-50% CO_2, along with small amounts of other gases and water vapour (Plugge, 2017). Biogas can be produced from a diverse range of feedstocks, including food waste, sewage sludge, animal manure and other types of organic materials.

In Europe, a significant portion of biogas production relies on energy crops (IEA, 2020). In contrast, biogas production in Sweden involves a low amount of energy crop feedstock (SEA, 2022), which can be explained by the absence of subsidies for energy crops, resulting in reduced profitability (Björnsson *et al*., 2016). Moreover, a majority (~65%) of the biogas produced in Sweden is upgraded to biomethane for the transport sector

(Energigas Sverige, 2022), while at global level biogas is primarily used for heat and electricity generation (IEA, 2020).

Biogas is as a versatile energy carrier, as it is capable of being stored for future use (Weiland, 2010). This flexibility aligns well with power systems with a large share of intermittent sources, such as wind and solar, as a complement during periods of low production. In addition, biomethane has almost identical characteristics to natural gas and can therefore be used directly in existing infrastructure (IEA, 2020). Besides energy, biogas production systems also yield a digestate that can be used as organic fertiliser, reducing the demand for synthetic fertiliser and adding carbon to the soil (Thomsen *et al*., 2013).

Grass is often proposed as an energy- and climate-efficient substrate for biogas production (Smyth *et al*., 2009; Tilman *et al*., 2009; Björnsson *et al*., 2016; Auburger *et al*., 2017). One reason for this is that grass cultivation is a thoroughly established agricultural practice that can be implemented in diverse conditions, without requiring adoption of new farming methods (Smyth et al. 2009). Other promising bioenergy crops with biomass potential are cover crops. In Sweden, cover crops are normally left unharvested due to their relatively low productivity, whereas they are harvested in other regions with a longer vegetation period (Molinuevo-Salces *et al*., 2013; Launay *et al*., 2022).

While agricultural land is a finite resource, opportunities exist for expanding agricultural activities by exploiting unused or underused arable land (Prade *et al*., 2017). For example, Börjesson (2016) estimated that around 100,000 ha of unused agricultural land in Sweden is available to increase domestic biomass production. This land resource is considered highly suitable for cultivation of energy crops, as it poses minimal short-term competition with food production. This means a lower risk of indirect land-use impacts, reducing the need for *e.g.* conversion of natural land (Tilman *et al*., 2009). Utilisation of this untapped potential aligns with the EU Renewable Energy Directive (RED), a regulatory framework designed to support increased implementation of renewable energy in the EU. The RED restricts the use of biofuels derived from food and feed crops to 7% of energy consumption in the road and rail transport sectors. However, an exemption is made for what is referred to as “low indirect land-use change-risk biofuels, bioliquids, and biomass fuels” (EU, 2018). Under this exemption, it is permissible to utilise abandoned or degraded land if the stipulated criteria outlined in Article 4 of Regulation 2019/807 are met, including the additional measures in Article 5 (EU, 2019). Another exemption is made for use of cover crops, provided that their cultivation does not influence the cultivation of other main food and feed crops within the cropping system, thereby triggering demand for additional land (EU, 2018).

In the recently published REPowerEU plan (EC, 2022), increasing biomethane production is recognised as pivotal to the aim of lowering EU reliance on Russian fossil fuels. This plan has set a target of 350 TWh across the EU by 2030 (EC, 2022). Current production in Sweden is 2.3 TWh (Energigas Sverige, 2022), but it has been suggested that production could be increased fourfold (SOU, 2019), which would contribute significantly contribute to increased energy security within the EU.

3.5 Life cycle assessment

3.5.1 Life cycle assessment methodology

Life cycle assessment methodology is a comprehensive approach that aims to evaluate the environmental impacts associated with a product throughout its entire life cycle, from production to disposal. There are multiple approaches to conducting LCA, but the most widely accepted framework is governed by the ISO LCA standard, specifically outlined in standards 14040:2006 and 14044:2006 (ISO, 2006a, 2006b). This framework comprises four main phases: (i) goal and scope definition, (ii) inventory analysis, (iii) impact assessment, and (iv) interpretation.

During the goal and scope definition phase, the purpose of the LCA is established, including the target audience and whether the results are intended to be compared with other products and services. The inventory analysis phase involves collecting data on all relevant inputs and outputs required to meet the defined goals and scope. In the impact assessment phase, the collected inventory data are aggregated to assess specific environmental impacts. Finally, in the interpretation phase, the results are analysed and contextualised, and suggestions for potential improvements are provided. All four phases of the LCA process are conducted iteratively, allowing for adjustments and refinements at any stage. Life cycle assessment is commonly employed to evaluate the environmental impact of agricultural products and is recognised by policymakers in both public and private organisations (Brandão *et al*., 2022b).

A crucial element of the LCA methodology is the functional unit chosen, which serves as the basis for assessing the environmental impact. The functional unit should represent the function or purpose of the system analysed and allow the environmental impact to be quantified relative to this function. The choice of functional unit should be clearly defined during the goal and scope phase of the LCA and can be either input-based (*e.g.* hectares of land) or output-based (*e.g.* MJ of biofuel produced). Determining the most appropriate functional unit for an assessment may not always be straightforward. In such cases, it is possible to include multiple units in the assessment to capture different aspects of the system (Klöpffer & Grahl, 2014). This approach helps to ensure comprehensive evaluation of the environmental impact within the LCA.

To increase the comparability and usability of LCA studies, the general practice is further specified in LCA frameworks such as the Environmental Product Declaration (EPD) and Product Environmental Footprint (PEF) (Brandão et al., 2022a). The EPD is compliant with ISO 14025, which is based on ISO 14040/14044 (EPD, 2021). The EPDs are performed on a voluntary basis by companies to declare the independent verified environmental information for products and services based on LCA. The PEF framework was developed by the European Commission to establish a unified methodology for quantitatively assessing and communicating the environmental impacts of products and services within the EU (EC, 2021). As of 2023, the framework is in a 'transition phase' before its adoption into EU policy. Renewable energy sources under RED must also meet sustainability criteria, including GHG reduction relative to fossil fuels, determined through strictly specified life-cycle climate impact calculations (EU, 2018).

3.5.2 Agricultural life cycle assessment

The LCA methodology was initially developed as a tool to quantify the environmental impact of industrial production. However, its application has since been expanded to encompass various systems, including agricultural production (*e.g.* Garrigues *et al*., 2012; Nemecek *et al*., 2015). When applied to agricultural systems, LCA has been criticised for its product-oriented approach, focusing solely on the provisioning outputs while neglecting other crucial services provided by sustainable agricultural systems (van der Werf *et al*., 2020). This has led to input-intensive agricultural systems being favoured in LCA studies per unit product, even though they are often associated with elevated environmental impacts per unit area (van der Werf *et al*., 2020). Furthermore, compared with industrial processes, agricultural systems interact to a larger degree with surrounding and supporting systems, *e.g.* the soil and adjacent landscapes. Consequently, the environmental impact of agricultural systems is influenced by temporal and spatial variations. An example of this is agricultural GHG emissions, which are highly dependent on spatial and temporal factors, such as climate, soil type and management practices (Miller *et al*., 2006). The spatial and temporal dimensions of the environmental impact of agriculture can be incorporated into every phase of LCA methodology. For instance, in the life cycle inventory data that represent the specific conditions prevailing at the investigated sites can be collected, while in life cycle impact assessment the environmental impact can be assessed for the specific spatial and temporal context. These aspects can also be included in the goal and scope phase and in the interpretation phase, *e.g.* when identifying the potential of environmental burden-shifting from one region to another (Patouillard *et al*., 2018).

Previous studies that have performed spatially explicit assessments of agricultural systems (*e.g.* Humpenöder *et al*., 2013; Hörtenhuber *et al*., 2014; Henryson et al. 2019) have highlighted the importance of spatial differentiation to obtain more relevant results than those of classic LCA studies. In addition, soil processes, such as SOC sequestration, need to be considered in LCA (Brandão *et al*., 2011; Vidal Legaz *et al*., 2017), since studies have shown that changes in SOC can have a substantial impact on the overall GHG balance of agricultural systems (*e.g.* Tidåker *et al*., 2014; Hammar *et al*., 2017; Yang *et al*., 2018).

Despite the criticisms directed at the method, agricultural LCA has become recognised as an effective tool for quantitatively assessing the resource use and environmental burdens of agricultural production, and has proven to be important in promoting environmental impact reductions for agricultural systems (Fan *et al*., 2022).

3.5.3 Climate impact assessment

Within the field of LCA, the most common metric used to assess the climate impact is global warming potential (GWP) (Cherubini & Strømman, 2011). It is determined by calculating the cumulative radiative forcing of a GHG emission in comparison with the cumulative radiative forcing of an equivalent amount of CO_2 over a specific time horizon, typically 100 years (Myhre *et al*., 2013). Since the emissions are relative to CO_2, the climate impact is given in CO_2-equivalents (CO_2-eq). This GWP calculation provides a standardised metric for quantifying the relative climate impact of different GHGs over a specified time horizon, facilitating comparisons of emissions throughout the system life cycle. There are pre-defined GWP characterisation factors for most GHGs, *e.g.* the factor for CH_4 is 34 and that for N_2O is 298 in a 100-year time horizon (GWP_{100}), according to Myhre *et al*. (2013). In 2021, these emission factors were updated to 29.8 and 27.0 for fossil and non-fossil CH_4, respectively, and 273 for N_2O (Forster *et al*., 2021). Another method for assessing the climate impact of GHG emissions is global temperature change potential (GTP). This method goes one step further and assesses the temperature change of the radiative forcing caused by the GHG emission at a specific point in time. This is achieved by applying radiative forcing calculation in combination with the temperature response to changes in the radiative forcing (Myhre *et al*., 2013). Both GWP and GTP are midpoint indicators, but GTP describes the climate impact a step further down the cause-effect chain of climate impact (Fuglestvedt *et al*., 2003). However, methods further down the cause-effect chain are associated with higher uncertainties than methods higher up, such as GWP.

One drawback with the GWP and the GTP approaches is that they give a static representation of the climate impact and do not include the timing of the emissions. This means that emissions which occur at different points in the life cycle are added together, even though the endpoint of the impact differs (Kendall, 2012; Cherubini *et al*., 2016). However, by investigating the time-dependent version of GWP and GTP, *i.e.* the absolute GWP (AGWP) and GTP (AGTP), the climate impact can be assessed dynamically throughout a specified analytical time horizon. This approach to assessing climate impact has been used previously in LCA studies to evaluate the climate impact, *e.g.* of bioenergy systems (Ericsson *et al*., 2013; Hammar *et al*., 2017). Unlike static GWP and GTP approaches, dynamic climate impact assessment entails establishing a life cycle inventory with temporal resolution, which leads to higher data demand. Levasseur *et al*. (2016) concluded that using multiple climate metrics and time horizons in LCA can enhance transparency by displaying uncertainties and impacts of metric selection.

Biogenic carbon dioxide emissions

Combustion of biofuels generates CO_2 emissions that are approximately equivalent to those from combustion of fossil fuels. Nevertheless, a common assumption is the notion of climate neutrality of the CO_2 emitted from bioenergy, which hinges on the cyclical pattern of the life cycle of biomass (Creutzig *et al*., 2015). In this cycle, plants capture carbon in the form of CO_2 from the atmosphere, via photosynthesis, and subsequently dies and decays or through other processes release their carbon content back to the atmosphere. Consequently, the sequestration and emissions of carbon associated with biomass may be seen as constituting a net-zero exchange with the atmosphere. However, this assumption has been criticised, as it does not consider the time lag between biogenic CO_2 emissions and sequestration (Brandão *et al*., 2013). Biogenic CO_2 emissions and sequestration are also associated with land use change, such as changes in the SOC stock (Creutzig *et al*., 2015). Accumulation and decomposition of SOC results in fluxes of temporally stored biogenic carbon, which affect the atmospheric CO_2 concentration and ultimately radiative forcing (Brandão *et al*., 2013).

Several methods have been proposed for incorporating temporary carbon storage and emissions into the climate impact assessment step of LCAs (Brandão *et al*., 2013). Another option is to use the dynamic climate impact method by assessing the AGWP or AGTP of a GHG emission throughout the selected time horizon. However, there is currently no consensus on which method should be used to incorporate these effects.

3.6 Agricultural models to simulate crop growth and GHG fluxes

Measurements of environmental emissions in cropping systems are often lacking due to high cost, time constraints and technical feasibility. The second best option is to use models. Agricultural models are used to model processes within the agricultural environment, such as crop growth and soil carbon and nitrogen fluxes. These models are increasingly being used in environmental planning and management for agriculture (Tonitto *et al*., 2018), and can be divided into two categories, statistical (also called empirical) and process-based. Statistical models are normally more straightforward and transparent, but because they rely entirely on the data used to derive the relationship, and in most cases they have a smaller geographical range (Smith *et al*., 2012). In contrast, process-based models can theoretically be applied to many combinations of geography, climate, cropping systems and management practices (Smith *et al*., 2012). In practice, however, their use is limited by lack of scientific knowledge of the modelled processes (Tonitto *et al*., 2018). This means that the results from process-based models must be carefully scrutinised.

3.6.1 Modelling soil organic carbon dynamics

The SOC balance is regulated by decomposition of soil organic matter, which is the microbial process whereby organic carbon is oxidised to CO_2 and inorganic substances are released into the soil environment, or incorporated into the microbial biomass (Lorenz & Lal, 2012). The process whereby the components in the decomposed material are transformed into inorganic substances is called mineralisation, while the process of assimilation of inorganic substances is called immobilisation (Ågren & Andersson, 2012). To date, the dominant paradigm of SOC decomposition has been that chemical recalcitrance regulates the decomposition of carbon in soils. Therefore, most soil carbon models are constructed around a type, or pool, of organic material that has an intrinsic decay rate. Labile organic matter that is decomposed is partly converted into CO_2 through microbial respiration and partly converted into a more stable pool (Schmidt *et al*., 2011). Some soil carbon models only simulate the soil carbon balance, such as the RothC model (Coleman & Jenkinson, 1996) and the Introductory Carbon Balance Model (ICBM) (Andrén & Kätterer, 1997). These models do not simulate crop growth and therefore data on carbon input are necessary to operate them. In contrast, dynamic agro-ecosystem models describe the interaction between crop growth, soil carbon and nitrogen dynamics, and environmental processes. Examples of such models are DNDC (DeNitrification DeComposition) (Li *et al*., 1992), DayCent (the daily time-step version of CENTURY) (Parton *et al*., 1998), and the Daisy model (a soil-plant-atmosphere model focusing on agro-ecosystems) (Abrahamsen & Hansen, 2000). Agriculture also affects CH_4 fluxes, mostly through rearing of livestock, but also through soil processes. Soils can act as a net sink or net source of CH_4, depending on moisture, soil nitrogen level and ecosystems. Native prairie and forests systems tend to be net consumers of CH_4 (Johnson *et al*., 2007).

3.6.2 Modelling soil N_2O emissions

The most important processes in emissions of N_2O from agricultural soils are biological nitrification and denitrification (Khalil *et al*., 2004). Nitrification is the process whereby ammonium (NH_4^+) is oxidised to nitrate (NO_3^-). The NH_4^+ enters the soil matrix through *e.g.* net mineralisation of organic nitrogen, deposition from the atmosphere or application of synthetic fertiliser (*Figure 2*). During nitrification, N_2O is formed as a by-product to varying degrees. Under anaerobic conditions, the NO_3^- in the soil can be reduced to nitrogen gas (N_2), which leads to losses of nitrogen from the soil. This process is called denitrification and is a four-step reaction in which N_2O is an intermediate (*Figure 2*).

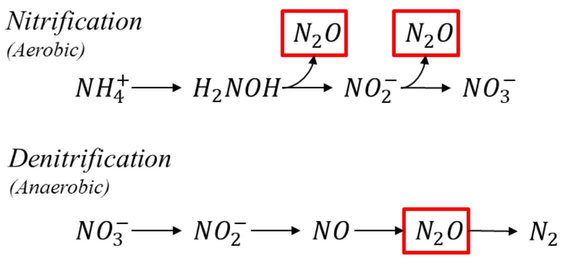


Figure 2. Production and consumption of different reactants in nitrification and denitrification.

Nitrous oxide is a very potent climate forcer, 273-298 times stronger than CO_2 over a 100-year perspective (Myhre *et al*., 2013; Forster et al. 2021), which means that even small emissions cause large radiative forcing. Estimates of soil N_2O emissions are associated with large uncertainties. The major reason for this is that the emissions show substantial temporal and spatial variations and that the underlying processes affecting the emissions are still not fully known (Butterbach-Bahl *et al*., 2013).

In LCA, the most common approach for estimating soil N_2O emissions is the IPCC Tier I approach, which is recommended by the IPPC when rigorously documented country-specific emission factors are lacking (IPCC 2006, 2019a). The main limitations with this approach are that: (i) it is site-generic and does not consider spatial variations between different types of soils and (ii) the emission factors are biased towards soils in mid-latitude regions, and are thereby not equally applicable to soils in the northern hemisphere (Rochette *et al.*, 2018). Process-based agro-ecosystem models can be used to estimate soil N_2O emissions for specific conditions and thereby increase understanding of N_2O emissions when assessing the life cycle impact of agricultural systems.

4. Materials and methods

4.1 System descriptions

In Papers I-IV, the climate effect of utilising grass and cover crops in Swedish agriculture was analysed. Papers I and II assessed the effect of grass cultivation and grass cultivation on unused arable land, while the other two papers examined the climate effect of incorporating grass in crop rotations (Paper III) and cultivation of oilseed radish as a cover crop (Paper IV). In Papers I and III, the assessment applied a cradle-to-farm-gate perspective, while in Papers II and IV, the scope extended to cradle-to-grave, including utilisation of the biomass to produce upgraded biogas for use as vehicle fuel and digestate for use as organic fertiliser.

4.1.1 Grass cultivation

In Paper I, grass cultivation was assessed at five different sites in Sweden, from Kungsängen in east-central Sweden to Tönnersa in the south-west. These sites represented different spatial properties, both in terms of soil and climate. Information about the sites is provided in Table 2. The weather data were collected for the 30-year period 1986-2015

Table 2. *Properties of the five sites studied in Paper I.*

Site	Karlslund	Klevarp	Kungsängen	Lanna	Tönnersa
Latitude	59.4	57.7	59.8	58.5	56.5
Mean temp (°C) 1986-2015	6.8	5.4	6.9	7.1	8.0
Soil texture	Clay loam	Sandy loam	Clay	Silty clay loam	Sandy loam
SOC (%)	2.6	1.7	6.0	2.0	1.5
Clay content (%)	29	2	57	33	3

The grass was cultivated in five-year rotations, starting with sowing and rolling in year 1 and concluding with ploughing in year 5 (*Figure 3*). Throughout the rotation, the grass was fertilised and cut twice a year. Two different fertiliser intensities were evaluated, 140 and 200 kg N ha^{-1}, in fertilisation scenarios denoted F1 and F2, respectively. The system was analysed over a 30-year period, *i.e.* six rotations, and the environmental impact was assessed per hectare (ha) and per Mg dry matter (DM) harvested biomass.

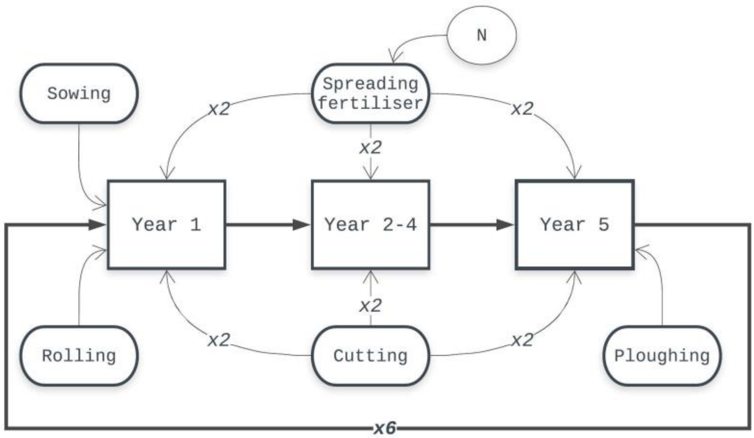


Figure 3. Schematic overview of the grass cultivation rotation assessed at the five sites in Paper I. The rotation began with sowing and rolling in the first year, and concluded with ploughing in year 5. During the rotation, the grass was fertilised and cut twice a year.

4.1.2 Grass-based biogas system

In Paper II, the grass cultivation system in Paper I was extended to include handling of the harvested biomass for biogas production. The study was conducted as a hypothetical case study for Uppsala Municipality in Sweden, where 3587 ha of land was reported to be in fallow in 2014. Information on current land use was obtained directly from the Swedish Board of Agriculture. All organic soils and fields smaller than 0.5 ha were omitted from the available area, which reduced the total to 3006 ha, at 1240 different sites. Approximately 90% of the soils were fine-textured and classified as either silty clay loam, clay loam, silty clay or clay (*Figure 4*). The initial carbon content in the soils exhibited considerable variation, ranging from 0.7% to 11.5%, with a median value of 2.2%. Weather data for a 10-year period (2007-2016) were applied in the model, looped through the 100-year study time.

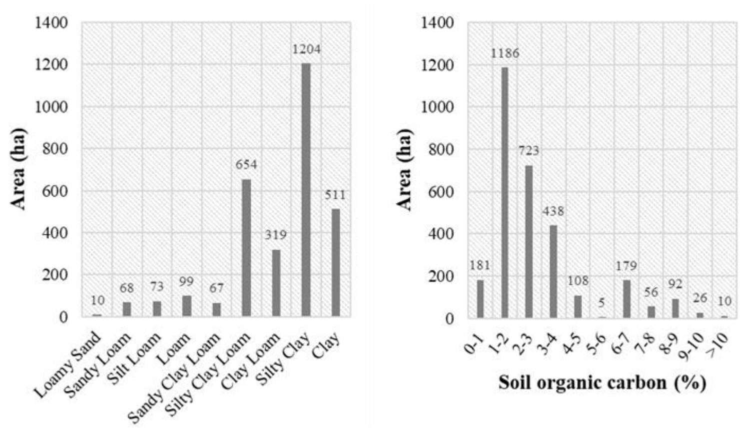


Figure 4. (Left) Soil texture characteristics and (right) initial soil organic carbon (SOC) content at the fallow sites (N=1240) in Uppsala Municipality assessed in Paper II.

The harvested biomass in each field was assumed to be transported to the biogas plant in the Municipality, where it was converted to upgraded biogas. The biogas plant was assumed to be located at the same site as the existing plant in Uppsala. The same management regime as in Paper I was assumed in each field (*Figure 3*). The impacts from the grass-based biogas system were compared with those of a business-as-usual scenario, which comprised fallow land at the investigated sites and using diesel instead of the upgraded biogas produced in the alternative system. The fallow was assumed to remain unmanaged throughout the study period, except for an annual autumn cut, at which the cut biomass was left in the field. The digestate produced was assumed to be used as organic fertiliser in winter wheat cultivation, replacing synthetic fertiliser in the business-as-usual scenario (*Figure 5*).

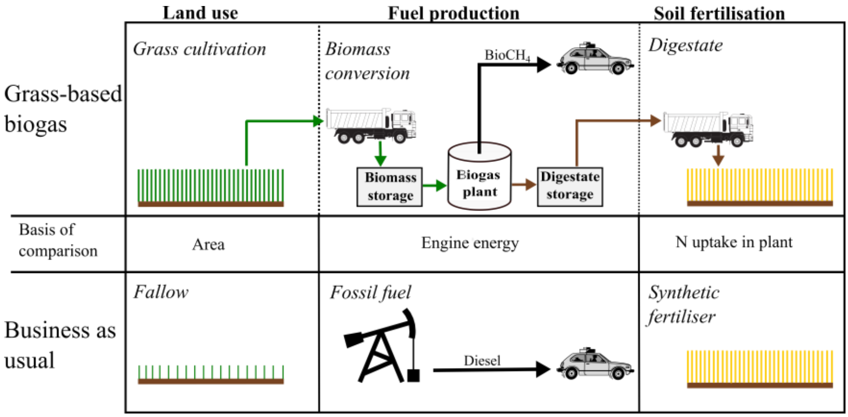


Figure 5. Schematic overview of the system investigated in Paper II. The grass-based biogas system consisted of the life cycle stages grass cultivation, biomass conversion and digestate utilisation as organic fertiliser. It was compared with a business-as-usual scenario incorporating the life cycle stages fallow land, fossil fuel and synthetic fertiliser usage. The net effect was calculated as the difference between the grass-based biogas system and the business-as-usual scenario.

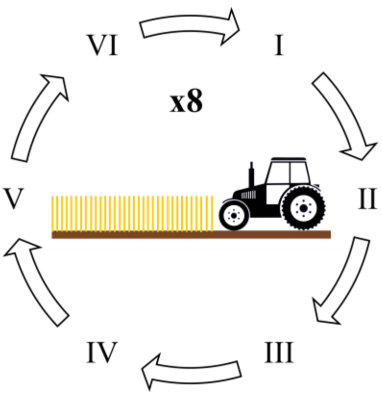
4.1.3 Grass in crop rotations

In Paper III, data from a long-term field experiment underway at three sites in Sweden were used to assess the climate impact of crop rotations with and without rotational grass. All three sites are located in southern Sweden, at Säby (59°49´N; 17°42´E), Lanna (58°20´N; 13°07´E) and Stenstugu (57°36´N/; 8°26´E) and have been in operation since 1969, 1965 and 1968, respectively. The characteristics of the sites are shown in Table 3.

Table 3. *Soil properties at the Säby, Lanna and Stenstugu sites in the Swedish long-term field experiment*

	Säby	Lanna	Stenstugu
Soil texture	Silty loam	Silt clay loam	Loam
Clay content (%)	23	35	21
Silt content (%)	53	49	31
Sand content (%)	20	12	45
Soil organic matter (%)	4.0	4.2	2.8

The crop rotation investigated comprised six crops, where the first four crops were identical (oilseed, winter wheat, oats, and barley) and the last two varied between the rotations: (i) two-year grass-legume mixture, (ii) two-year pure grass and (iii) spring wheat and fallow, referred to hereafter as (i) *Mixed*, (ii) *Grass* and (iii) *No-Grass* rotations. These rotations were investigated under two different nitrogen application regimes, *High N* and *Low N*. The assessment was performed using data from eight full rotations, *i.e.* a total of 48 years (*Figure 6*).



Rotation	*Mixed*	*Grass*	*No-Grass*
I	Oilseed crop	Oilseed crop	Oilseed crop
II	Winter wheat	Winter wheat	Winter wheat
III	Oats	Oats	Oats
IV	Barley	Barley	Barley
V	Clover/grass ley 1	Grass ley 1	Spring wheat
VI	Clover/grass ley 2	Grass ley 2	Fallow

Figure 6. Schematic overview of the system analysed in Paper III. Three different crop rotations (*Mixed*, *Grass* and *No-Grass*) were investigated over eight full rotations, *i.e.* over 48 years.

4.1.4 Energy cover crop

In Paper IV, data on biomass growth and nitrogen content were obtained from two field experiments established in 2018 and 2019 in southern Sweden, where oilseed radish was cultivated as an unfertilised cover crop.

The cover crop was established at different times after the preceding crop. The experiment measured growth of both aboveground and belowground biomass. Three different alternative scenarios involving cover cropping were compared with a reference scenario without cover cropping. In this reference scenario, the field was assumed to be left bare between the preceding and subsequent main crop, while the alternative scenarios comprised cover crop cultivation with: (i) the cover crop left in the field over winter and ploughed under and incorporated into the soil in spring (*Incorporation* scenario), (ii) aboveground cover crop biomass harvested in autumn by mowing (*Mowing* scenario) and (iii) aboveground and belowground biomass harvested in autumn by uprooting the cover crop (*Uprooting scenario)* (*Figure 7*). In the *Mowing* and *Uprooting* scenarios, the harvested biomass was assumed to be transported to a co-digestion biogas plant located 50 km away from the field, where it was converted into upgraded biogas, which was used to substitute for fossil diesel fuel in the transport sector. The digestate produced was assumed to be returned to the field and used as organic fertiliser.

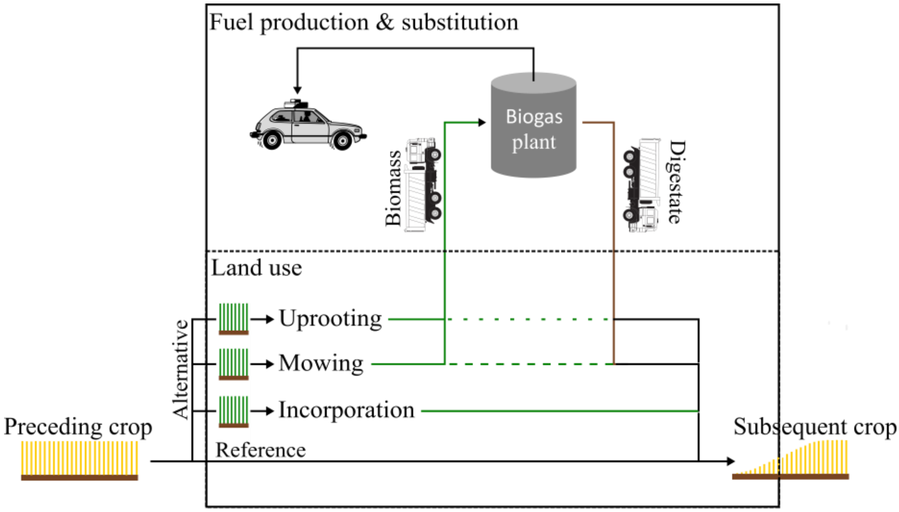


Figure 7. Overview of the different scenarios investigated in Paper IV. The reference scenario involved bare soil between the main crops, while the alternative scenarios introduced cover cropping (oilseed radish). In the *Incorporation* scenario, the cover crop was ploughed under and incorporated into the soil in spring. In the *Mowing* scenario, aboveground biomass was harvested by mowing in autumn. In the *Uprooting* scenario, aboveground and belowground biomass were harvested in autumn. The harvested biomass from the *Mowing* and *Uprooting* scenarios was converted into upgraded biogas and used to substitute for fossil diesel fuel.

4.2 Agricultural modelling for crop growth and soil carbon and nitrogen fluxes

4.2.1 DeNitrification DeComposition (DNDC) model

The DNDC model was employed in Papers I and II to generate data for the life cycle inventory regarding biomass growth, SOC dynamics and soil-borne N_2O and CH_4 emissions. The model is based on equations derived from classical laws of physics, chemistry and biology, as well as empirical laboratory observations (Li *et al.*, 2006). It was initially developed to simulate carbon and nitrogen fluxes in agricultural soils (Li *et al.*, 1992), but has since been refined and updated by numerous researchers worldwide to suit specific research purposes, leading to the development of different versions of the model (Gilhespy *et al.*, 2014). In the work in this thesis, a

Canadian version of the model (DNDC-CAN) specifically tailored and validated for Canadian conditions, which are similar to the cool-weather conditions prevailing in Sweden, was utilised. This version has undergone refinements, *e.g.*to more accurately simulate crop biomass growth (Kröbel *et al*., 2011), soil temperature (Dutta *et al*., 2017), and evapotranspiration (Dutta *et al*., 2016). Recently, DNDC-CAN was expanded to simulate perennial regrowth after harvests in subsequent years (He *et al*., 2019). The model has been employed in previous LCA studies to simulate the impacts of agricultural systems (Goglio *et al*., 2014, 2018b).

In Papers I and II, the DNDC-CAN model was provided with site-specific data regarding management regime, soil properties, climate conditions and location. Data on specific soil properties such as porosity, density, field capacity and wilting point were acquired using a pedotransfer model created by Saxton & Rawls (2006). Analysis of the model fit to observed biomass growth data was conducted in Paper I.

The model set-up from Paper I was used in Paper II, but with grass cultivation assumed to take place on selected fallow land within Uppsala Municipality. These sites were given specific properties by interpolating measurements in the Municipality regarding SOC, clay, sand and silt content, as well as pH levels. Interpolation was performed using Geographic Information System (GIS) programming.

4.2.2 Introductory Carbon Balance Model (ICBM)

In Paper IV, the SOC sequestration potential of cultivating oilseed radish under different management strategies was assessed using the Introductory Carbon Balance Model (ICBM), a relatively simple soil carbon model for simulating change SOC stocks in agricultural soils (Andrén & Kätterer, 1997). The model theoretically divides the soil into two carbon pools, a young pool and an old pool. When carbon enters the soil, via cover crop residues or digestate from anaerobic digestion, it initially enters the young pool. From there, carbon is either transferred to the old pool or returned to the atmosphere as CO_2 through oxidation. Further oxidation of carbon occurs in the old pool, but at a slower rate. The fraction of the carbon input that enters the old pool is described by the humification coefficient, which varies depending on the source of the carbon input. The remaining carbon in both the young and the old pool at a specific time was calculated as a measure of carbon sequestration in the system. In Paper IV, humification coefficients from Bolinder *et al.* (2018) were used. Since there are no peer-reviewed humification coefficients for digestate, the coefficient for farmyard manure was used. Additional belowground carbon input in the form of exudates, root hairs and fine roots, which were not included in the sampled root material, was included in calculations. This additional input was assumed to constitute 65% of the sampled root biomass, based on results from Bolinder *et al.* (2007)

4.2.3 IPCC Guidelines for National Greenhouse Gas Inventories

The IPCC guidelines for National Greenhouse Gas Inventories provide methodologies for estimating N_2O emissions from agricultural activities, including nitrogen fertilisation, livestock manure management and crop residue management (IPCC, 2019a). According to these guidelines, direct soil N_2O emissions induced by nitrogen inputs can be estimated by multiplying the amount of nitrogen by an emission factor. In Paper I, soil N_2O emissions modelled with the DNDC model were compared with emissions estimated according to IPCC Tier I guidelines and the site-specific empirical approach developed by Rochette *et al.* (2018). The IPCC approach was used in Paper III, with the default parameters, and also in Paper IV, but in that case specific emission factors for cover crop cultivation and digestate, based on literature data, were used.

4.3 Life cycle assessment

The system boundaries for the studies in Papers I, II, III and IV are shown in *Figure* 3, 5, 6, and 7, respectively. In Paper I, the environmental impact was assessed per ha and per Mg DM of harvested grass biomass. In Paper II, three different units were applied: (i) ha of agricultural land, (ii) MJ of upgraded biogas produced and (iii) all investigated fields in Uppsala Municipality. In Paper III, an entire crop rotation was assessed, and consequently the environmental impact was assessed per ha for the rotation, and per unit yield and per Mg DM of the entire rotation, but also as cereal units (CU), a concept developed by the German authorities with the aim of making agricultural production more comparable. In this approach, harvested crop biomass in Mg DM is converted into cereal units using a conversion factor based on the animal feeding value of the crop normalised to a reference crop (barley) (Brankatschk & Finkbeiner, 2014). Cereal unit have been used in previous

LCA studies as a functional unit or to allocate the environmental burden from a cropping system to different crops in the rotation (Brankatschk & Finkbeiner, 2015; Goglio *et al*., 2018a; Henryson *et al*., 2019). The conversion factors used in Paper III are shown in Table 4.

Table 4. *Conversion factors used to calculate amount of cereal units in Paper III*

Crop	Conversion factor
Oilseed	1.30
Winter wheat	1.04
Oats	0.84
Barley	1.00
Spring wheat	1.04
Grass*	0.61

*Referred to as "hay" in Brankatschk & Finkbeiner (2014).

4.3.1 Life cycle impact assessment

Climate impact

After the life cycle inventory, the next step in LCA is to estimate the environmental impact caused by these emissions. In Papers I and II, the climate impact was analysed using the GWP method and the dynamic climate impact model using AGTP, as described in section 3.5.3. In Papers I and II, GWP was calculated using emission factors from Myhre *et al.* (2013), while in Papers III and IV, updated emission factors from Forster *et al.* (2021) were applied. All significant fluxes of GHGs, including CO_2, CH_4 and N_2O, throughout the life cycle (see sections 4.1.1-4.1.4) were considered.

Biogenic CO_2 emissions resulting from combustion of the biogas produced were not considered in the climate impact assessment in Papers II and IV. However, the climate impact of changes in SOC stocks was assessed by dynamically analysing annual changes using AGTP and with the static GWP metric. In the GWP calculations, mean annual changes in SOC stocks, modelled over 30- and 100-year periods, were included in Papers I and II, respectively. Measured average change in SOC over 48 years was incorporated into the GWP calculation in Paper III. In Paper IV, the modelled SOC change remaining after 100 years, induced by the cover crop system, was included in the GWP calculation.

Paper II specifically compared the climate impact of the biogas produced to the impact of the fossil diesel fuel alternative to calculate GWP reduction in the system:

$$GWP\ reduction = \frac{(GWP_F - GWP_B)}{GWP_F} \qquad (1)$$

where GWP_F is GHG emissions from fossil diesel, including production and utilisation, and GWP_B is GHG emission from the grass based-biogas system in Paper II, excluding the fossil fuel substitution.

Eutrophication

In Papers I and III, the eutrophication impact was evaluated using the CML method (Guinée, 2002), a commonly employed characterisation method for eutrophication in LCA. However, this method has certain limitations as it is site-generic, placing the impact indicator at the point of the emissions without considering the fate of eutrophying emissions. In addition, it does not account for whether the recipient is nitrogen- or phosphorus-limited, thus treating all nitrogen and phosphorus discharges as potentially eutrophying. Hence, it does not fully capture the complexity of eutrophication, especially in Sweden, where the end-recipient is the surrounding Baltic Sea, the world's largest brackish water basin. The Baltic Sea exhibits varying degrees of nitrogen and phosphorus limitation in different sub-basins (SEPA, 2006). To address these complexities and complement the CML method, a site-specific approach was employed in Paper I. This method, developed by Henryson *et al.* (2018), provides emissions factors tailored for different regions in Sweden, offering a more comprehensive assessment of marine eutrophication.

Energy resource depletion

In Paper II, the efficiency of the grass-based biogas system was evaluated using energy ratio as presented by Djomo *et al.* (2011):

$$Energy\ ratio = \frac{E_{Out}}{E_{in}} \qquad (2)$$

where E_{Out} is the biogas produced from the system, and E_{in} is the primary energy input to the system. Energy produced and utilised within the system boundary, such as biogas used for heating in the anaerobic digestion process, was not included in the energy ratio.

In Paper III, energy resource depletion was evaluated using the abiotic depletion potential for energy carriers (Van Oers & Guinée, 2016).

5. Results and discussion

5.1 Climate impacts

5.1.1 Grass cultivation

In Paper I, the GHG fluxes with the strongest influence on the overall GHG balance were soil N_2O emissions, emissions from fertiliser manufacturing and changes in the SOC stock. The higher fertiliser rate (F2) resulted in lower GHG emissions per unit yield but generally led to higher emissions per unit area, although the difference was small (*Figure 8*). The higher fertiliser rate contributed to higher GHG emissions per ha from fertiliser manufacturing and soil N_2O emissions. However, these elevated emissions were partially offset by greater SOC sequestration. Overall, the climate impact of grass cultivation showed larger variation between sites than between fertiliser rates. The fine-textured soil at Kungsängen displayed the highest GHG emissions per yield, particularly at the lower fertiliser rate (F1), whereas emissions were lower for the coarser-textured soils with lower initial SOC content (Klevarp and Tönnersa).

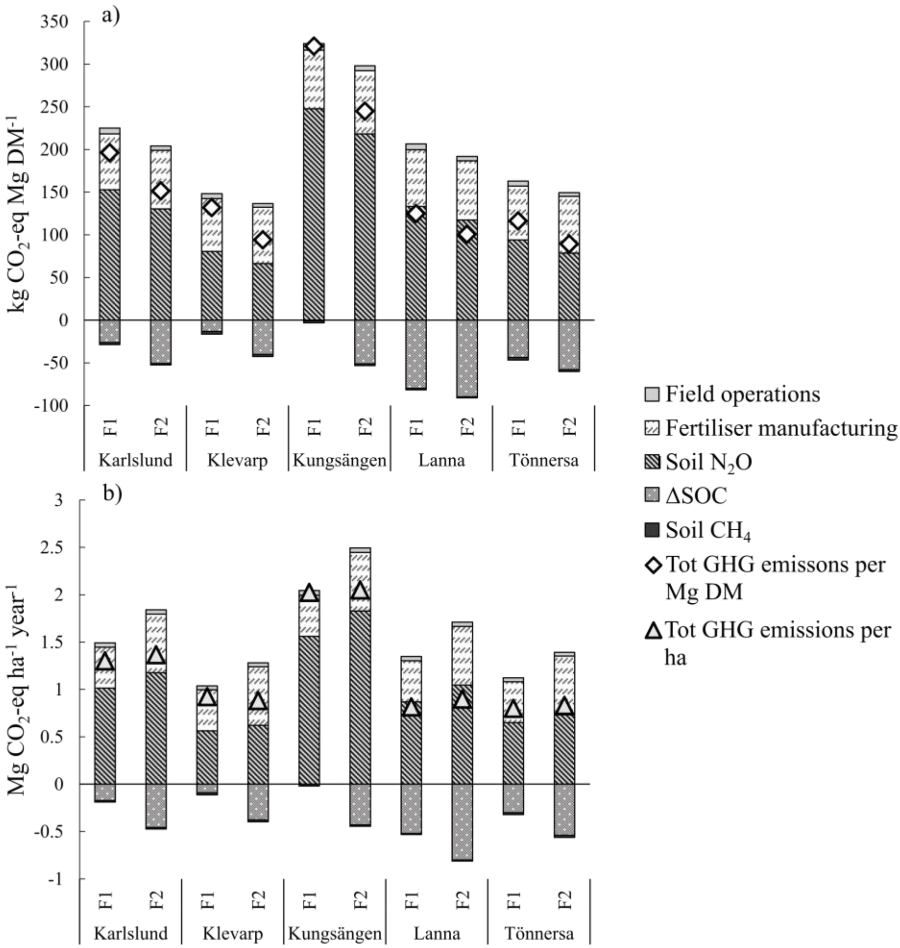


Figure 8. Greenhouse gas (GHG) emissions from grass cultivation at the five study sites in Paper I, expressed as (a) kg CO_2-eq per Mg dry matter (DM) yield and (b) Mg CO_2-eq per ha and year, for fertiliser rates F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}). The GHG emissions are divided between field operations, fertiliser manufacturing, soil nitrous oxide (N_2O) emissions (direct and indirect), changes in soil organic carbon (SOC) stock and soil methane (CH_4) emissions.

In Paper II, a reference scenario was employed to analyse the net effect of introducing grass cultivation on fallow land in Uppsala Municipality. The results showed large variation between sites in the region (*Figure 9*). However, as in Paper I, the most influential GHG fluxes from land use were from fertiliser production, soil-borne N_2O emissions and changes in the SOC stock. The GHG emissions from field operations were greater than in Paper I, due to assumed chopping of the grass biomass to facilitate subsequent anaerobic digestion.

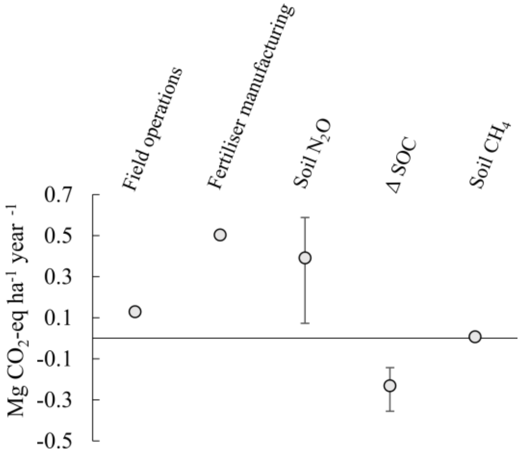


Figure 9. Net greenhouse gas (GHG) emissions (Mg CO_2-eq per ha & year) resulting from grass cultivation on fallow land in Uppsala Municipality in Paper II. Circles represent emissions from the median soil in the region, while bars indicate the highest and lowest emissions in the emissions categories field operations, fertiliser manufacturing, soil nitrous oxide (N_2O) emissions, soil organic carbon (SOC) change and soil methane (CH_4) emissions.

5.1.2 Grass in crop rotation

In Paper III, the climate effect of integrating two-year grasses, either pure grass (*Grass*) or a grass-legume mixture (*Mixed*), into crop rotations under two different fertiliser regimes was investigated. The results showed that GHG emissions per ha were greater for the higher fertiliser rate (*High N*) across all sites. In the rotation without grass (*No-Grass*), which included a one-year fallow, inputs were lower than in the grass rotations, resulting in lower GHG emissions from inputs and soil N_2O emissions. However, the grass rotations required fewer field operations and the observed SOC depletion was generally greater in the *No-Grass* rotation (*Figure 10*).

The grass rotations led to higher yield per ha for the annual crops in the first four years of the rotations under the lower fertiliser regime (*Low N*), an effect that was particularly evident in the rotation including legumes (*Mixed*).

A lower effect of grass inclusion on the yield of annual crops was observed under the higher fertiliser regime (*High N*). Total yield, *i.e.* including all crops in the rotation, was also higher for the grass rotations, mainly due to the one-year fallow without biomass harvest in the *No-Grass* rotation. This resulted in lower GHG emissions per unit yield for the grass rotations compared with the rotation without grass. The lowest GHG emissions per unit yield were observed for the *Mixed* rotation under the lower fertiliser regime. This was primarily attributable to lower input of nitrogen fertiliser per unit yield, which reduced the GHG emissions, both soil N_2O emissions and emissions from production of fertiliser.

Adoption of cereal unit as the functional unit resulted in a smaller difference in GHG emissions per unit yield between the grass rotations and the *No-Grass* rotation, compared with using DM yield as the functional unit. This was explained by the higher cereal unit conversion factors assigned to the annual crops that dominated the *No-Grass* rotation, while the grass biomass was considered to have a lower conversion factor.

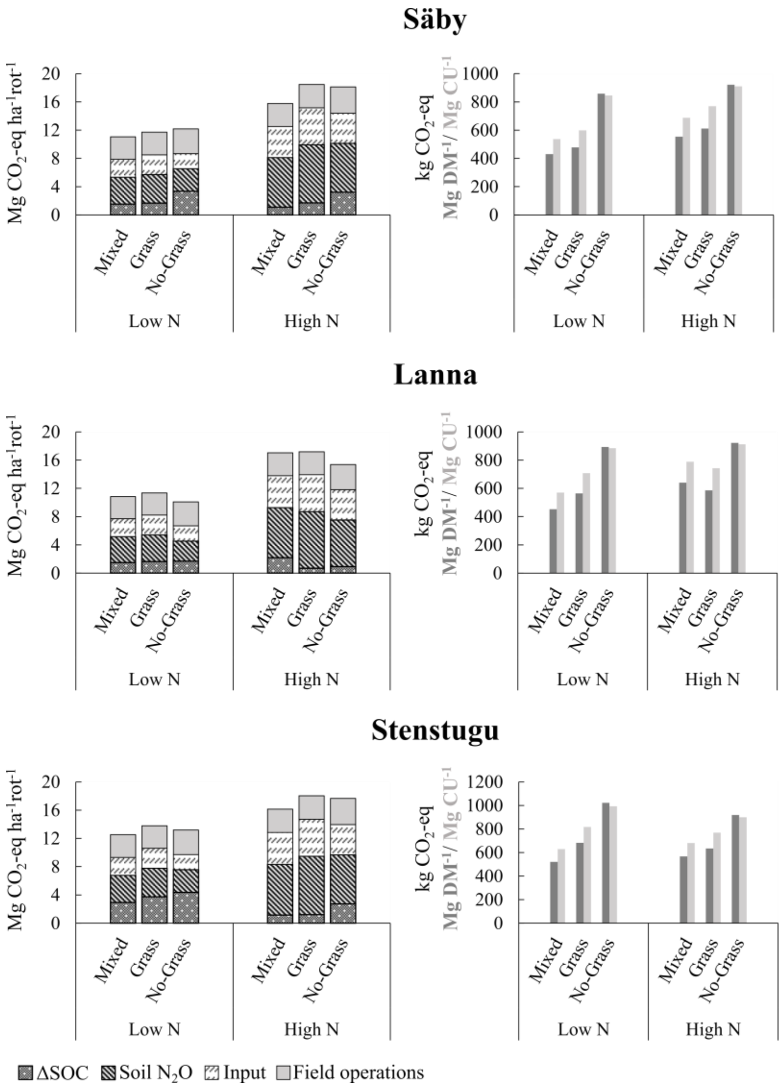


Figure 10. Greenhouse gas (GHG) emissions per ha and rotation (left) and per Mg dry matter (DM) and Mg cereal unit (CU) (right) in the Mixed, Grass and No-Grass treatments assessed at the sites Säby, Lanna and Stenstugu in Paper III.

5.1.3 Grass-based biogas system

The system boundary in Paper II was expanded to include use of the harvested biomass as feedstock to produce upgraded biogas. The biogas produced was assumed to replace fossil diesel in the transport sector and the digestate was assumed to be used as an organic fertiliser to replace synthetic fertiliser. *Figure 11* shows mean annual GHG emissions from the system, incorporating all fallow land considered in the analysis. In the *Land use* life cycle stage, fertiliser manufacturing and soil N_2O emissions resulted in the greatest increase in regional emissions compared with the reference scenario, while SOC sequestration reduced the emissions. In the *Fuel production* life cycle stage, the largest GHG emissions were primarily due to CH_4 losses during biogas production and digestate storage (*Figure 11*). A large emission reduction was also achieved in this life cycle stage through diesel fuel substitution. Within the *Soil fertilisation* life cycle stage, the most substantial potential for GHG emission reductions was through increased SOC storage and substitution of synthetic nitrogen fertiliser manufacture with digestate application. Overall, the system studied gave a mean net GHG emissions reduction of 9950 tonnes CO_2-eq per year, corresponding to 4.4% of Uppsala Municipality’s annual GHG emissions from the transport sector (Uppsala Municipality, 2023).

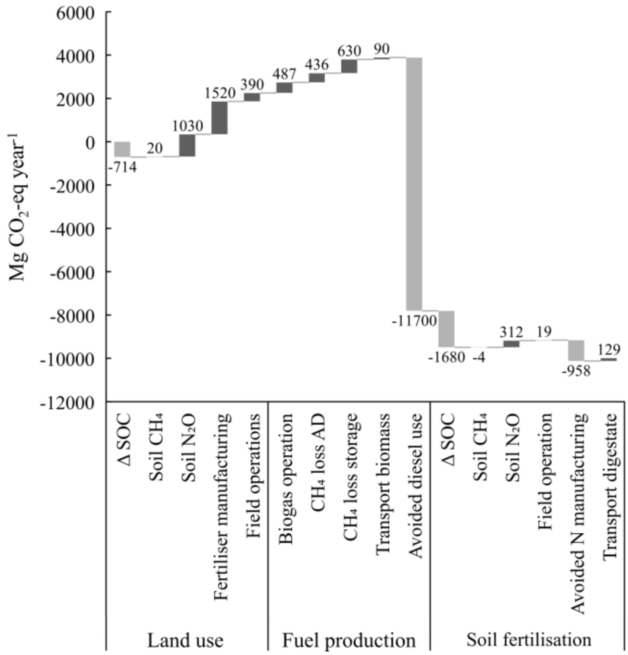


Figure 11. Mean annual cumulative net greenhouse gas (GHG) emissions from the grass-based biogas system in Uppsala Municipality, using all available fields (N=1240, 3006 ha) (Paper II). Emissions, evaluated using Global Warming Potential over a 100-year timeframe (GWP_{100}), are categorised into the life cycle stages: Land use, Fuel production and Soil fertilisation. The results are presented as Mg CO_2-eq.

On excluding substitution of fossil diesel, utilisation of upgraded biogas produced on all fallow land included in the study had GWP of 10 g CO_2-eq MJ^{-1}. This corresponded to an 85% reduction in GWP compared with use of diesel fuel. However, due to considerable variation in emissions between sites (*Figure 9*), the reduction potential ranged between 102% and 79%, contingent upon the location within the region where grass cultivation took place. *Figure 12* demonstrates the GWP reduction per MJ of the grass-based biogas produced relative to diesel depending on the proportion of total fallow land area utilised. For instance, when only the top 10% of best-performing land (in terms of climate impact) was assumed, the GWP reduction increased to 95%.

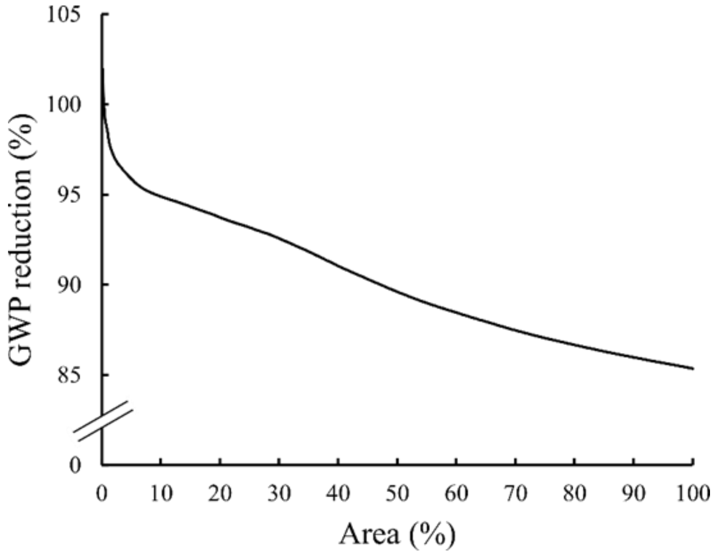


Figure 12. Global Warming Potential (GWP_{100}) reduction from the grass-based biogas system in Paper II in relation to proportion of total available fallow land area included in the study.

To complement the static GWP metric, a dynamic assessment of the temperature response was conducted (*Figure 13*). Within the grass-based biogas system, the biomass conversion life cycle stage made the greatest contribution to the short-term climate impact. However, over time, the significance of the grass cultivation stage increased, because the primary GHG emitted during biomass conversion was CH_4, a relatively short-lived climate forcer, resulting in a diminishing rate of climate impact over time from this stage. When considering the overall effect of the system, the impacts of the land use and soil fertilisation life cycle stages more or less offset each other over the 100-year time horizon.

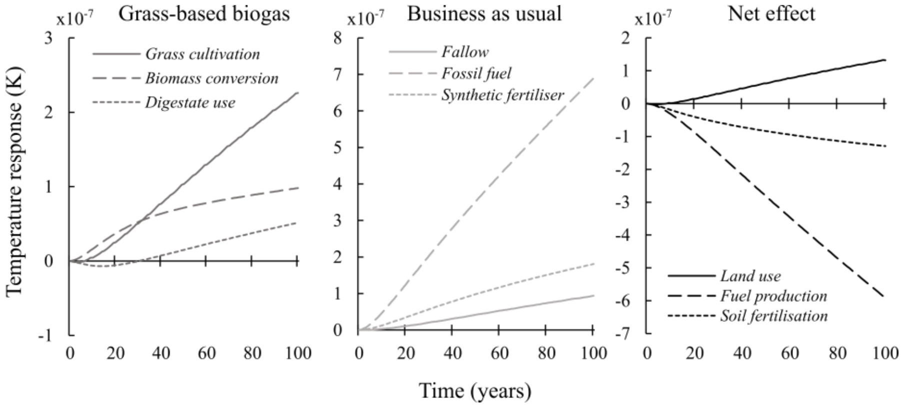


Figure 13. Temperature response, in degrees kelvin (K), when using all available fields (N =1240, 3006 ha) in Uppsala Municipality in Paper II. Climate impact from (left) the grass-based biogas system and (centre) the business-as-usual scenario, and (right) the net effect, i.e. the difference between the grass-based biogas system and business-as-usual scenario. The impact from the grass-based biogas system is divided into the life cycle stages grass cultivation, biomass conversion and digestate use, while that in the business-as-usual scenario is divided into the life cycle stages fallow, fossil fuel and synthetic fertiliser use. The net effect is divided into impact from land use (grass cultivation - fallow), fuel production (biomass conversion - fossil fuel) and soil fertilisation (digestate - synthetic fertiliser).

5.1.4 Energy cover crop

The results in Paper IV indicated that the climate impact of a cropping system can be reduced by introducing cover crops and that harvesting and utilising the cover crop biomass to produce upgraded biogas can result in greater mitigation potential (*Figure 14*). Similarly, to the results in Papers I-III, soil N_2O fluxes made an important contribution to the GHG emissions from the system assessed in Paper IV. As no inputs in the form of fertiliser and biocides were applied to the cover crop, climate impacts from production of these were not included. In the *Uprooting* scenario, cover crop cultivation reduced the need for synthetic nitrogen fertiliser input compared with the reference scenario. However, the other alternative scenarios resulted in a deficit in the system's nitrogen balance, which was compensated for by adding synthetic nitrogen fertiliser. Cover crop cultivation resulted in additional soil carbon input compared with the reference scenario, leading to an increase in SOC stock.

The highest GHG emissions in the *Fuel Production and Substitution* life cycle stage were from transport of biomass to the biogas plant and transport of digestate back to the field. The *Mowing* and *Uprooting* scenario generated 9 and 18 GJ upgraded biogas per ha, respectively, which could be used to replace diesel fuel production and consumption corresponding to GHG emissions of 0.6 and 1.2 Mg CO_2-eq, respectively. Overall, the results showed that the *Uprooting* scenario had the greatest mitigation potential, but lower SOC sequestration potential, compared with the other cover crop scenarios.

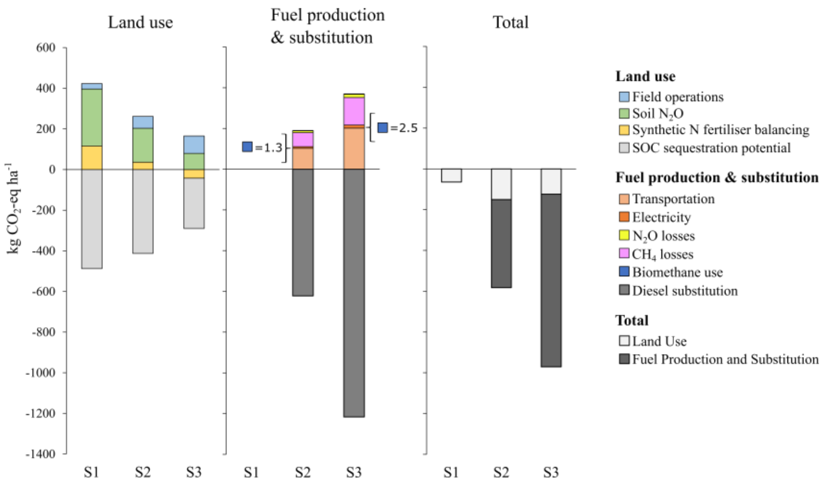


Figure 14. Greenhouse gas (GHG) balance in kg CO_2-eq per ha for the alternative scenarios involving cover crop cultivation in Paper IV (Incorporation (S1), Mowing (S2) and Uprooting (S3)) in relation to the Reference scenario, without cover crops. The emissions are divided into the life cycle stages Land use (left), fuel production and substitution (centre), and total GHG balance of the system (right).

The timing of cover crop establishment had a large influence on the GHG balance of the system, with earlier establishment generally resulting in greater biomass production, and consequently less nutrient leaching and greater potential for SOC sequestration and diesel fuel substitution. This led to a greater mitigation potential with early establishment of the cover crop, while late establishment led to lower mitigation potential. However, the early cover crop establishment may be unattractive to farmers, as it also requires early harvesting of the preceding crop in the rotation. Harvest in early July is not suitable for most crops under Swedish conditions, but is possible for some, such as early-harvested peas (“green peas”). Early establishment of cover crops could also be possible when immature cereals are harvested for fodder. Harvesting the belowground biomass in addition to aboveground biomass (*Uprooting*) increased the yield per ha and the climate change mitigation of the system. However, this approach can increase the risk of dirt, such as sand and grit, being introduced into the biogas reactor, which may cause operational difficulties (Steffen *et al*., 1998).

5.2 Soil organic carbon balance

In Papers I and II, the change in SOC through introducing grass cultivation on unused land was investigated. In Paper I, the effect was simulated over 30 years, while in Paper II it was simulated over 100 years. In Paper I, the SOC stock increased at all sites, with a greater effect at the higher fertiliser rate (F2, 200 kg N ha^{-1}) compared with the lower rate (F1, 140 kg N ha^{-1}) (*Figure 15*). However, low carbon sequestration potential was observed for the Kungsängen soil under the F1 fertiliser regime. Soil organic carbon sequestration potential also showed wide variation between sites, as the value over the 30-year period varied between 0 and 4 Mg C ha^{-1} for the F1 fertiliser regime and between 3 and 6.5 Mg C ha^{-1} for the F2 fertiliser regime. According to Bolinder *et al*. (2017), grass cultivation in Sweden could be expected to achieve SOC sequestration of about 0.56 and 0.085 Mg C ha^{-1} y^{-1} in topsoil and subsoil, respectively. However, these values correspond to the net effect compared with cultivation of annual cereal crops, while the results presented in *Figure 15* only illustrate the gross effect. The net effect is, of course, heavily dependent on the chosen reference scenario.

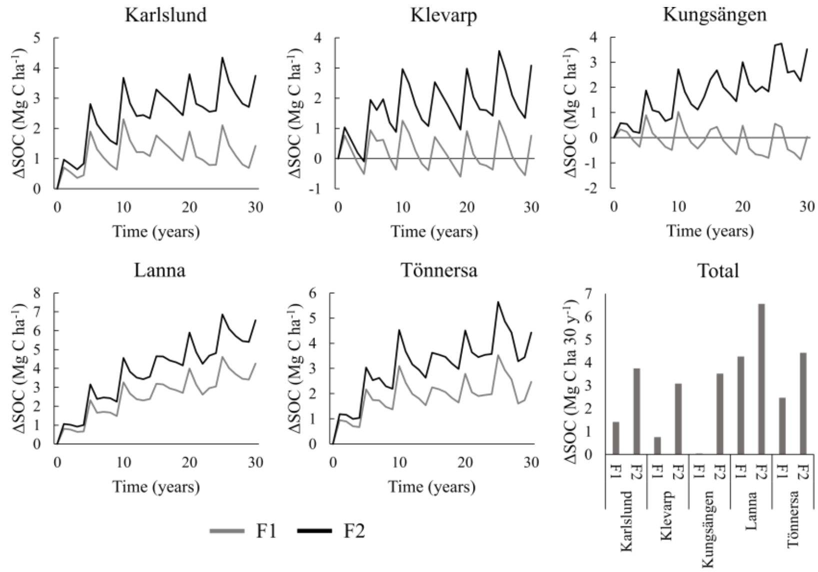


Figure 15. Simulated soil organic carbon (SOC) balance for the five sites investigated in Paper I over a 30-year time horizon. The bottom right graph shows total SOC change over the 30-year period. The grey line represents fertiliser rate F1 (140 kg N ha^{-1}), while the black line represents fertiliser rate F2 (200 kg N ha^{-1}).

In Paper II, the grass was assumed to be cultivated on reported fallow land in Uppsala Municipality, in a system was over a 100-year time horizon. The sequestration potential was calculated as the difference between grass cultivation and the reference land use, which was assumed to be green fallow with low productivity. Soil organic carbon balance again showed large spatial variation within the region, with introduction of grass cultivation leading to increased SOC stock at some sites and depleted SOC stock at others (*Figure 16*). A similar pattern was found for the fallow sites, with the difference that the SOC increase was lower and the depletion higher in relation to the grass cultivation sites. This difference between grass cultivation and the reference land use resulted in a net increase in SOC at all sites, meaning that over a 100-year horizon grass cultivation resulted in greater SOC stock at the sites investigated compared with the green fallow land use. The spatial variability in the net effect on SOC change was reduced due to counterbalancing of variations between grass cultivation and fallow land. The dynamic aspect of SOC balance was more apparent in Paper II, which adopted a 100-year perspective, than in Paper I, which had a study period of 30 years.

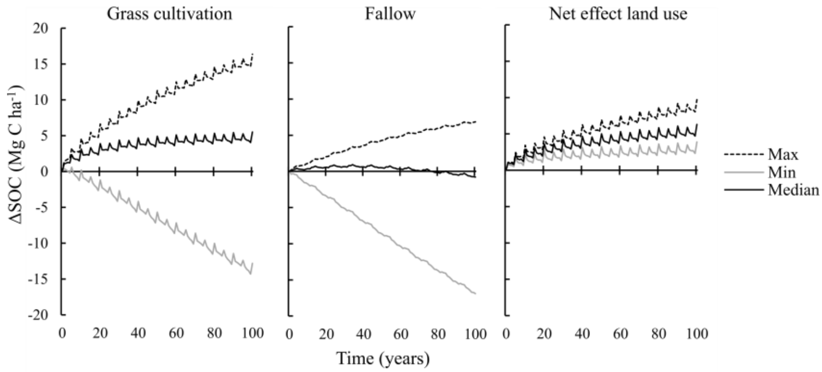


Figure 16. Simulated soil organic carbon (SOC) balance over 100 years for all sites included in Paper II (N = 1240, 3006 ha) for (left) grass cultivation and (centre) fallow land, and (right) net effect of changing land use from fallow to grass cultivation. The dashed black line represents the 95th percentile (max), the grey line the 5th percentile (min) and the black line the median.

The modelled changes in SOC observed in Papers I and II showed the highest correlation to initial SOC stock and soil clay content. In Paper II, the correlation coefficient for to these parameters (r-value) was -0.79 and 0.50, respectively. This indicates that higher initial SOC content resulted in lower SOC sequestration potential, while higher clay content led to higher SOC sequestration potential. Similar findings have been reported in other studies (*e.g.* Kätterer *et al*., 2012; Poeplau *et al*., 2015). The reason for this is that soils with high initial carbon content are typically near their carbon saturation level, resulting in lower capacity for carbon sequestration, and that a high clay content affects the decomposition rate by limiting the physical availability of organic material to soil decomposers (Li *et al*., 1992).

The SOC effect of utilising digestate as fertiliser was analysed in Paper II. The study compared the effect on SOC stock of digestate application in winter wheat cultivation in relation to application of synthetic fertiliser. The results revealed increased SOC stock with digestate application, while use of synthetic fertiliser led to considerable depletion in SOC stock, which resulted in a large net increase in SOC stock for the digestate application scenario. Compared with the land use stage, utilisation of digestate as a fertiliser gave a greater net increase in SOC stock within the soil fertilisation life cycle stage (*Figure 17*)

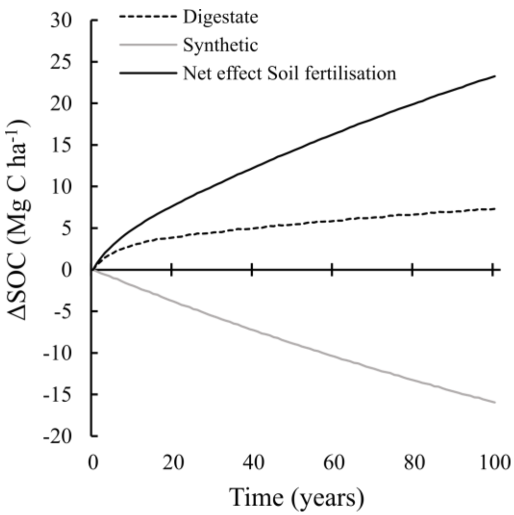


Figure 17. Simulated soil organic carbon (SOC) balance in winter wheat cultivation (Paper II), comparing use of biogas digestate as fertiliser (dashed line), application of synthetic fertiliser (grey line) and the net effect (black line), which represents the difference between digestate and synthetic fertiliser. Each scenario was modelled over a 100-year time horizon. The SOC balance simulation reflects average conditions in the region.

In contrast to Papers I and II, where the change in SOC stock was based on modelled values, Paper III used measured values of SOC when including grass (pure grass and legume-grass mix) in crop rotations. The assessments was performed for eight full six-year crop rotations, *i.e*. 48 years. The results showed on average SOC depletion at all sites and for all rotations and fertiliser scenarios (*Figure 18*). However, the results demonstrated large variation between the replicated plots, which is indicated by the error bars in *Figure 18*. The average changes in SOC stock were influenced by both inclusion of grass in the crop rotation and the rate of fertiliser application. Soil organic carbon depletion was generally lower in the rotation including grass (*Grass* & *Mixed*) and at the higher fertiliser rate (*High N*). However, at the Lanna site, the *Mixed* grass rotation under the *High N* fertiliser regime gave the greatest SOC depletion, which contradicted the pattern observed at the other sites. Under certain conditions, increased carbon inputs, such as increased crop residues as a result of increased fertilisation, may stimulate microbial activity and accelerate degradation of existing carbon in the soil (Blagodatskaya *et al*., 2011). These phenomena, known as priming effects, are often used to explain cases where increased carbon inputs lead to enhanced carbon decomposition (Poeplau *et al*., 2015). However, the mechanisms underlying priming effects and their interconnections are not yet fully understood (Liu *et al*., 2020). In Paper III, there was no additional evidence to suggest that these mechanisms was responsible for the greater SOC depletion in the mixed grass rotation with the higher fertiliser rate at Lanna. On analysing the average SOC content over the entire duration of the long-term field experiment, including all sites, it was observed that the rotation without grass inclusion had significantly lower carbon stock than the grass rotations.

The decline in SOC stock seen across all sites in Paper III may be attributable to the two-year grass representing an insufficient proportion of grass in the crop rotation (Jarvis *et al*., 2017; Zani *et al*., 2021). Another potential reason for SOC depletion may be the initial SOC content (Kätterer *et al*., 2012). Details of the former land use at each site were unavailable, but it is plausible that the experimental set-up involved fewer perennial crops, as a higher proportion of grass was more common in the past.

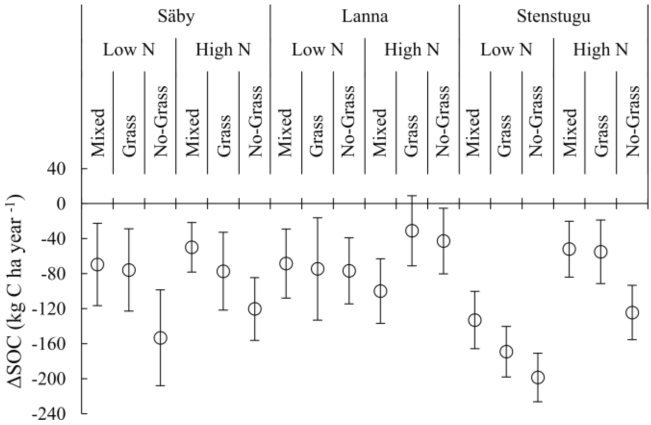


Figure 18. Change in soil organic carbon (SOC) in each treatment and site in Paper III. The mean value for each treatment is marked with a circle. Error bars indicate the 95% confidence interval.

In Paper IV, the ICBM model was used to estimate the potential SOC effect of introducing cover crops into a cropping system in southern Sweden under different management practices. The largest SOC effect was observed for the *Incorporation* scenario (*Figure 19*), where the cover crop was left unharvested, resulting in a SOC increase of 0.13 Mg C ha^{-1} after 100 years. In comparison, the *Mowing* and *Uprooting* scenario resulted in an increase of 0.11 and 0.07 Mg C ha^{-1}, respectively. These results indicate that although aboveground biomass was harvested in the *Mowing* scenario, the difference in the SOC effect was low when the digestate was returned to the same field. This aligns with the concept that the most easily degradable material during anaerobic digestion would also have decomposed quickly in the soil if the biomass had not been harvested. However, there is currently no reliable scientific evidence to refute or support this. The *Uprooting* scenario, where both aboveground and belowground biomass were harvested, exhibited a significantly lower SOC effect. Moreover, the potential SOC effect of including the cover crops in cropping system depended on the frequency of cover crop occurrence (*Figure 19*). For instance, in the *Incorporation* scenario, cover crop intervals of 3, 5 and 10 years yielded a SOC effect of 6.2, 3.4 and 1.7 Mg C ha^{-1}, respectively.

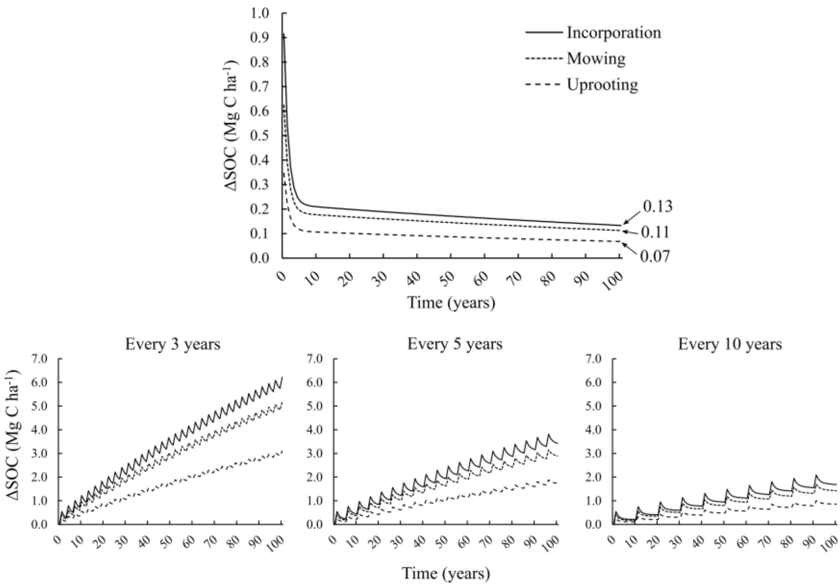


Figure 19. (Upper panel) Effect on soil organic carbon (SOC) stock of cover crop cultivation over a 100-year period, expressed in Mg C ha^{-1}, for the three management scenarios compared in Paper IV: Incorporation, Mowing and Uprooting. The diagrams show the amount of SOC remaining after 100 years following one-year of an oilseed radish cover crop, which was used in life cycle assessment to evaluate the climate effect corresponding to potential SOC sequestration. The lower panels show the effect of integrating the cover crop in a cropping system at intervals of 3, 5 and 10 years.

5.3 Soil-borne nitrous oxide emissions

Estimation of soil-borne N_2O emissions in Papers I and II was performed using the DNDC model, whereas the approach outlined in the IPCC guidelines for national greenhouse gas inventories (IPCC, 2019a) was adopted in Papers III and IV. To further assess the variation in soil N_2O emissions depending on the different estimation approaches, the results obtained in Paper I using the DNDC approach were compared with those obtained using another site-specific method developed by Rochette *et al.* (2018) and the site-generic method provided by IPCC (2019a). The two site-specific methods showed substantial variation between sites (*Figure 20*). The

DNDC model projected the highest N_2O emissions from the clay-rich soil in Kungsängen, whereas the Rochette *et al.* approach predicted the highest emissions from the field in Lanna, characterised by the lowest soil sand content. Both site-specific models consistently projected the lowest N_2O emissions from the sandy loam soils at Klevarp and Tönnersa. In contrast, the variation observed between sites using the site-generic IPCC approach was only due to differences in crop residue amounts, resulting from the different growth rates at different sites.

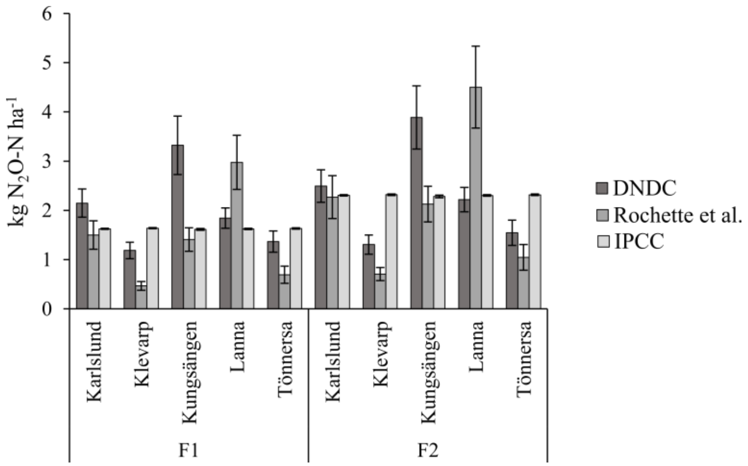


Figure 20. Mean nitrous oxide (N_2O-N) emissions at the five study sites in Paper I under the fertiliser rates F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}), as estimated using three different modelling approaches (DNDC, Rochette et al. and IPCC).

The variation in N_2O emissions across different spatial conditions and over time was further assessed in Paper II, where the DNDC model was used to estimate emissions from grass cultivation and green fallow at the sites in Uppsala Municipality. The N_2O emissions induced by grass cultivation were generally higher than those from green fallow, resulting in elevated regional soil-borne N_2O emissions from implementing the grass-based biogas system (*Figure 21*). The primary factor influencing emissions was soil pH, exhibiting a negative correlation, followed by initial SOC content, which displayed a positive correlation. The temporal variation in emissions was attributable to weather patterns, including precipitation and temperature.

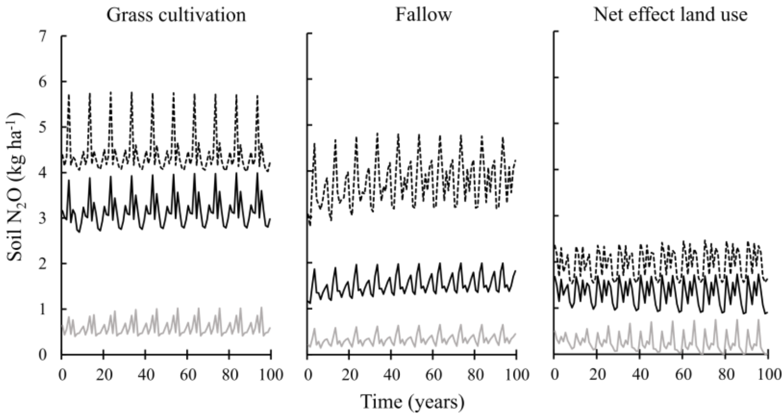


Figure 21. Simulated annual soil nitrous oxide (N_2O) emissions for a span of 100 years, representing all sites investigated in Paper II (N=1240, 3006 ha), for (left) grass cultivation and (centre) fallow land, and (right) net effect of transitioning the land use from fallow to grass cultivation. The dashed black line represents the 95th percentile (max), the grey line the 5th percentile (min) and the black line the median.

In Paper IV, new emission factors were developed for the IPCC approach to estimate the soil-borne N_2O emissions from the cover crop system assessed. The emission factor for N_2O emissions induced by cover crop residues was determined using literature data pertaining to unfertilised oilseed radish as a cover crop in Scandinavian conditions. The mean value for this emission factor was found to be 0.0153 kg N_2O-N per kg N in cover crop residues. However, large variation between studies was observed, with the 95% confidence interval ranging between 0.0214 and 0.0091 kg N_2O-N kg N^{-1}. In estimation of digestate-induced N_2O emissions, the emission factor used was derived from a study conducted by Launay *et al.* (2022) where the mean value was found to be 0.0052, with max and min values of 0.019 and 0.0008, respectively. The results in Paper IV showed that the scenario with the highest N_2O emissions induced by cover crop cultivation was that where the crop was left unharvested and incorporated into the soil (*Incorporation*), while in the *Mowing* and *Uprooting* scenarios the N_2O emissions induced by cover crop cultivation were lower (Figure 22). However, some of the reduced emissions in these scenarios were offset by elevated emissions from application of digestate back to the field. Furthermore, the cover crop scenarios were estimated to reduce nitrogen leaching compared with the reference scenario, which led to reduced indirect N_2O emissions. This also resulted in a decrease in emissions from avoided use of synthetic fertiliser. However, most of the synthetic nitrogen savings from the avoided leaching were more than offset by sequestration of nitrogen in soil organic matter through the increased SOC stock. This resulted in additional synthetic nitrogen being required in the *Incorporation* and *Mowing* scenarios. In contrast, there was a small reduction in demand for synthetic nitrogen in the *Uprooting* scenario, which was due to less nitrogen being lost within the system boundaries when harvesting the cover crop, and lower levels of nitrogen being sequestered in soil organic matter due to the lower SOC sequestration potential.

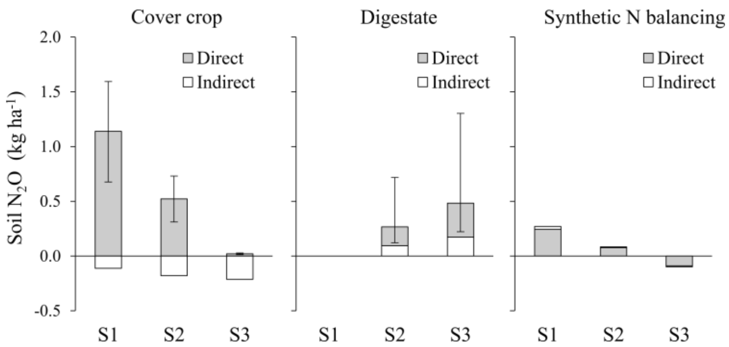


Figure 22. Estimated nitrous oxide (N_2O) emissions, in kg ha^{-1}, from the cover crop system assessed in Paper IV, calculated using the IPCC approach with specific emission factors for oilseed radish cover cropping and digestate application, in the Incorporation (S1), Mowing (S2) and Uprooting (S3) scenarios compared with a reference scenario with no cover crop cultivation. The emissions are categorised into cover crop-induced (left), digestate-induced (centre) and emissions resulting from the reduced need for synthetic nitrogen fertiliser (right).

5.4 Other impacts

In Paper I, the eutrophication impact was assessed using two methods: (i) the CML method developed by Guinée (2002) and (ii) the site-specific method developed by Henryson *et al.* (2018). According to the CML approach, eutrophication potential was highest for the Klevarp and Tönnersa sites, where simulated leaching rates were also highest. The Henryson approach identified Tönnersa as having the highest eutrophication impact, due to its proximity to recipient waters.

In Paper III, the eutrophication potential and energy resource depletion of the crop rotation treatments were assessed. The impact on these indicators followed the same trend as seen in the GWP assessment, with the lowest impact per CU for the *Mixed* rotation under the lower fertiliser (*Low N*) regime. Nitrogen emissions, predominantly via soil leaching but also through air-borne emissions, resulted in the highest eutrophication potential. At the site in Lanna, phosphorus leaching also made an important contribution to the eutrophication potential. Under the *Low N* regime, the majority of total energy depletion originated from field operations, while under the *High N* regime a larger proportion of energy depletion came from agricultural inputs. This was because the higher fertiliser rate caused greater total depletion, both per ha and per CU, compared with the lower rate.

The calculated energy balance of the system in Paper II showed that the primary energy input was greatest in the biomass conversion subsystem, where most of the energy was used for upgrading, compression, pumping and stirring in the biogas reactor. The second largest primary energy input was in the grass cultivation subsystem, where most of the energy input was used for fertiliser manufacture. In total, the primary energy input was 48 TJ y^{-1} and the energy output was 167 TJ y^{-1}, which resulted in an energy ratio of 3.5.

6. General discussion

6.1 Grass and cover crops for biogas production and climate change mitigation

6.1.1 Effects on agricultural system resource efficiency

One objective of this thesis was to investigate the climate effect of grass and cover crop cultivation for utilisation as feedstock in biogas production. The grass cultivation system assessed in Papers I and II could be considered to represent an intensification of the agricultural landscape, as unused land was re-introduced into agricultural production. This involved increased use of agronomic inputs, predominantly in the form of fertilisers and diesel fuel for field operations. However, an associated benefit of grass cultivation is that it could help improve soil fertility and thereby increase potential crop production in a region. Furthermore, during dry spells, which are expected to become more frequent in the future with a changing climate (IPCC, 2023), the grass produced could represent a valuable fodder reserve, thereby increasing resilience to climate-related contingencies.

In Paper III, including grass in crop rotations had a positive effect on yield of the subsequent annual crop under the lower fertiliser (*Low N*) rate. This was especially evident in the *Mixed* rotation with the legume-grass mixture. The same effect was not observed for the higher fertiliser rate (*High N*), which suggests that the positive effects on biomass provisioning gained via diversification were partly offset by elevated nitrogen fertiliser rate. Similar findings have been presented by *e.g.* MacLaren *et al.* (2022). Overall, the climate impact per cereal unit was lowest for the crop rotation including the two-year *Mixed* grass at the lower fertiliser rate. This was due to relatively high yield in relation to the amount of agronomic inputs, mainly in terms of synthetic nitrogen fertiliser. This shows that introduction of mixed grass-legume crops into crop rotations can be used to increase resource efficiency and mitigate the climate impact. However, the lower fertiliser rates resulted in higher land occupation, in terms of area needed per cereal unit produced, compared with the higher fertiliser rate. Land occupation is an important factor to consider, since increased demand for agricultural land could lead to clearing of new land, which is considered one of the primary contributors to climate and biodiversity impacts of agricultural systems (Foley *et al*., 2011). The risk of this type of indirect land use change was considered low in this thesis, however, as Sweden has a large amount of under-utilised agricultural land (Olofsson & Börjesson, 2016). In addition, there is great potential to free up currently used agricultural land by utilising the biomass produced more efficiently, *e.g.* by people converting to a more plant-based diet (Mottet *et al*., 2017). However, in order to increase inclusion of grass in crop rotations, there needs to be a market for grass biomass and one option is to promote its use as feedstock in biogas systems.

In Paper IV, cover crop cultivation was found to reduce nutrient leaching from the cultivation system. In addition, conversion of the biomass to digestate increased the fertilisation value, and it could thereby be used to substitute for synthetic fertiliser. However, the increased SOC stock led to sequestration of nitrogen in the soil, resulting in both the *Incorporation* and *Mowing* scenarios causing a deficit in the nitrogen balance compared with the *Reference*. As a result, more synthetic nitrogen fertiliser was needed in those scenarios to compensate for the deficit. The influence of SOC on the nitrogen balance of the systems is consistent with the underlying principle that, stoichiometrically, nitrogen must also be proportionally incorporated when carbon is sequestered (van Groenigen *et al*., 2017).

6.1.2 Soils as a carbon sink

The systems assessed showed potential to sequester carbon, although the studies revealed large spatial variations. Grass cultivation on former fallow land and inclusion of grass in crop rotation both resulted in losses of SOC under certain conditions. However. the losses were larger in the scenarios without grass cultivation, with fallow land (Paper II) and for the crop rotation without grass (Paper III). These findings highlight the importance of considering the SOC balance when conducting climate impact assessments of agricultural systems. However, the mitigation potential of using agricultural soils as a carbon sink is finite, since all soils become carbon-saturate over time (Smith, 2014). This means that soils with a high SOC content may have low capacity for further carbon sequestration, as shown in Papers I-III. In a study by Englund *et al.* (2023) examining the effect of large-scale grass inclusion in crop rotations in Europe, Sweden was among the countries that showed lower potential for SOC sequestration, while sequestration potential was highest for parts of Europe where the largest SOC losses have occurred historically. Furthermore, soils do not provide permanent storage, which means that carbon sequestration is reversible and the carbon stored in the soil may at any moment be re-emitted into the atmosphere, *e.g.* due to changes in the management scheme or in environmental conditions, which typically occurs faster than SOC build-up (Smith, 2005).

The impermanence of the carbon sink is especially important to consider when establishing carbon credit schemes, such as the proposed carbon farming mechanism in the EU (Radley *et al*., 2021). In such schemes, the idea is that carbon offset via SOC sequestration can be used to compensate for GHG emissions elsewhere (Paul *et al*., 2023). The benchmark for permanence is usually 100 years (Radley *et al*., 2021). This time period is rather short compared with the fossil CO_2 emissions for which compensation is made, which cause climate perturbations for thousands of years (Archer *et al*., 2009). Another problem with carbon crediting schemes is to ensuring additionality (Paul *et al*., 2023). In best practice, it should be proven that the carbon credit is generated via measures that would not have taken place without the presence of a crediting scheme. Since there are many positive aspects of sufficient SOC content from a provisioning point of view, and many farmers already include practices that build SOC, it may be difficult to prove additionality in these cases (Thamo & Pannell, 2016).

The carbon and nitrogen cycles in agricultural soils are closely connected, which means that changes in fluxes in one cycle will ultimately change fluxes in the other. One example is that carbon sequestered in soils will ultimately also sequester nitrogen, based on stoichiometric conditions. According to van Groenigen *et al.* (2017) approximately 100 Tg nitrogen per year would be required to achieve the “4 per 1000” target, *i.e.* to increase the carbon content in soils by 4 per mille per year. Production of the fertiliser required for this would come with additional GHG emissions. Increased nitrogen application is also associated with increased emissions of N_2O, which according to Guenet *et al.* (2021) has led to overestimation of the mitigation potential of SOC sequestration. This is confirmed by the results in Papers I-III, where the higher fertiliser regimes resulted in increased SOC sequestration rates, but also elevated N_2O emissions.

6.1.3 Substitution effects

Papers II and IV included the environmental benefit of replacing diesel with upgraded biogas produced from biomass harvested in the systems assessed, which was found to result in large avoided GHG emissions in these systems. In addition, the digestate produced was used to substitute for synthetic fertilisers. Biogas production per ha was considerably greater in the grass-based biogas system, where on average 55 GJ ha^{-1} was produced, compared with the cover crop biogas system, where 9 and 18 GJ ha^{-1} was produced in the *Mowing* and *Uprooting* scenario, respectively. However, avoided emissions are highly uncertain and are affected by a number of conditions and assumptions. In the investigations performed in this thesis, it was assumed that the biogas generated was upgraded for use as vehicle fuel. This assumption was based on prevailing conditions in Sweden, where vehicle fuel is the most common area of use for biogas (Energigas Sverige, 2022). However, other studies have suggested that it is more efficient from a GHG perspective to utilise the biogas in combined heat and power plants to produce electricity and heat (Shinde *et al.*, 2021). This is highly dependent on the GHG emissions associated with the substituted electricity and heat, especially the latter, which requires a sufficient heat demand in the region (Hijazi *et al.*, 2016). This highlights the large spatial and temporal variability in the substitution effect since biogas has different substitution values depending on where it is produced and where it is used. The substitution effect is also dependent on when the biogas is used, as the displaced source may shift over time, for instance as a result of more extensive use of renewable energy or of technology transitions such as electrification of the vehicle fleet (Creutzig *et al.*, 2015). This is especially relevant when interpreting the results in Paper II, where a static reference scenario was adopted throughout the 100-year time horizon. It would perhaps be more realistic to assume that the mitigating effect of the biogas produced declined over time, due to a higher proportion of renewables in the energy mix.

The notion of bioenergy serving as a transitional energy source on the path to electrification may be oversimplified, as there are areas of application where biofuels still offer the most realistic option for decarbonisation, such as in the maritime and aviation sectors (ETC, 2018). Furthermore, the flexibility of bio-based energy production makes it a suitable component in energy systems with large shares of renewable weather-dependent energy sources (Sepulveda *et al*., 2018). An addition advantage of biofuels over other renewables is that they are often relatively easy to integrate into existing infrastructure, which means that transition away from fossil fuels could occur earlier with bio-based fuels than when waiting for a full systemic shift to electrification. At the same time, too much focus on fuel switching from fossil fuels to biofuels could strengthen lock-in mechanisms and further delay a systemic shift to clean electrification (Olsson & Bailis, 2019).

6.2 Life cycle assessment as a tool to quantify the climate effects of bioenergy systems

6.2.1 Time- and space-dependent LCA

The GWP method was applied in all papers (I-IV) in this thesis. While this method may not offer distinct benefits compared with other available methods, it can be deemed more accessible for communication purposes and its widespread adoption has facilitated comparisons of results with other methods. For example, GWP is the metric applied for climate impact calculations in RED (EU, 2018). In Papers I and II, the GWP method was complemented by the dynamic climate impact method, using AGTP, which has been employed in previous studies, *e.g.* by Ericsson *et al.* (2013). This method considers the relevance of different types of GHG emissions over time, thereby providing a more accurate representation of the climate impact, including a more accurate representation of temporary carbon storage. The dynamic climate impact method requires temporal resolution of the life-cycle inventory, which increases the data demand. This makes it suitable to combine with agricultural modelling if data from long-term field experiments are not available.

Biogenic CO_2 emissions from biogas combustion were not included in the GWP calculations in Papers II and IV, since the energy crops studied were associated with short time lags between sequestration and emissions.

However, for energy crops with longer rotations, such as energy forests, these fluxes should be considered.

The large variations between sites and over time seen in the results for the different systems are typically not included in LCAs (Chaplin-Kramer *et al.* 2017). In this thesis, fluxes related to land use and land use change were observed to have considerable effects on the results and were highly influenced by the spatial variation in the systems assessed. In fact, the GHG emissions from grass cultivation in Papers I and II were more influenced by spatial and temporal variations than by the amount of nitrogen fertiliser applied. The GHG fluxes that caused the greatest variations between sites were soil N_2O emissions and changes in SOC stocks. A previous study by Patouillard *et al.* (2018) argued that there is a lack of guidance in agricultural LCA on prioritisation of efforts regarding the inclusion of spatial variation. Moreover, Fan *et al.* (2022) concluded that, due to the high cost associated with collection of large amounts of data, methodologies need to be developed based on limited data requirement to reflect the spatial variation in agricultural LCA. The results presented in this thesis show that when aiming to include spatial variability, soil carbon and N_2O fluxes are of the utmost importance. By examining the spatial differences in climate impacts within a region, as in Paper II, site-specific LCA can be used to strategically allocate land resources, giving preference to areas best suited for utilisation.

6.2.2 Modelling of agricultural soils

The DNDC model used in Papers I and II has preciously been used successfully to simulate agricultural processes in many different locations (Gilhespy *et al.*, 2014). In a study comparing different methods for estimating soil-borne CO_2 and N_2O emissions, Goglio *et al*. (2018b) found DNDC to be the model that yielded emissions estimates most consistent with measurements for N_2O. An advantage of employing such agro-ecosystem models, as opposed to more simplistic carbon models, is their capacity to concurrently simulate crop growth and nitrogen-carbon fluxes, thereby encompassing the interrelationships between these. However, use of simpler carbon models, such as ICBM, in conjunction with the IPCC approach employed for estimating soil N_2O emissions in Paper IV, provides an advantage in terms of user-friendliness and accessibility.

The ICBM model and the decomposition sub-model in DNDC follow the same principle, where the organic material is divided into different carbon pools with different decomposition rate. The ICBM model considers two carbon pools (Andrén & Kätterer, 1997), while the DNDC is a complex and considers four major pools, which in turn are divided into two or three sub-pools with different decomposition rates depending on pool size, soil temperature, soil moisture, clay content and nitrogen availability (Li *et al*., 1994). These types of models are important to understand and predict SOC changes, but the accuracy of the prediction is dependent on model validation (Le Noë *et al*., 2023). To further increase the validity of these models, more long-term field experiments on SOC dynamics under different conditions are needed. Estimation of SOC stock changes was long based on the notion that organic matter stabilisation is solely influenced by molecular structure, *i.e.* the inherent recalcitrance of organic matter to decomposition. However, this notion has been challenged and consideration of physical protection for soil carbon has been emphasised (Schmidt *et al*., 2011; Dungait *et al*., 2012). This implies that other soil mechanisms are equally or more important, such as micro-environmental conditions that limit the access (or activity) of decomposer enzymes, including hydrophobicity, soil acidity or sorption to surfaces (Schmidt *et al.*, 2011). These mechanisms need to be further investigated and better incorporated into models to improve predictions of SOC dynamics.

Modelling soil N_2O emissions is associated with large uncertainties. In LCA, the most common method used for estimating N_2O emissions is the IPCC Tier I approach, which the IPCC recommends only when data are lacking due to inability of the model to account for spatial heterogeneity in soil and climate properties (IPCC, 2006). The recent incorporation of spatial dimensions into IPCC Tier I methodology, achieved by disaggregating emission factors by climate region (IPCC, 2019a), represents a stride forward in addressing spatial variation in N_2O emissions, albeit at limited resolution. In Paper I, the two site-specific models (DNDC, Rochette *et al*.) were compared with the IPCC Tier I approach. The DNDC model gave the highest estimated N_2O emissions for clay-rich soils in the dataset and lower estimates for more coarse-textured soils. Estimates produced using the site-specific approach developed by Rochette *et al.* (2018), based on measured data in Canada, also showed a high correlation to soil texture. However, with this approach N_2O emissions were highest for the soil with the lowest sand content, while emissions were lowest for the same sites as with the DNDC model. Further research is needed to better understand the mechanism behind

N_2O emissions and the relationship with soil, crop and climate variables. Based on the results in Paper I, it is not possible to give a general recommendation on which model is best.

6.2.3 Interlinkages between crops in cropping systems

Agricultural LCA typically focuses on a single crop and hence frequently overlooks interlinkages between crops cultivated within the same cropping system (Nemecek *et al.* 2015; van der Werf *et al.* 2020). To address these interlinkages, one strategy is to adopt a systems LCA approach, which shifts the focus toward evaluating and contrasting entire production systems rather than isolating a single system output (Goglio *et al*., 2018a).

In Paper III, the effect of incorporating grass into crop rotations on yield of other crops within the same rotation was included in the assessment. Furthermore, the value of the different crops included in the rotation was considered using the cereal unit as the functional unit for the entire crop rotation, *i.e.* adopting the systems LCA approach. The same approach has been used in earlier studies (Prechsl *et al*., 2017; Henryson *et al*., 2019). The cereal unit may also be used to allocate the environmental burden between crops in the rotation, which is more relevant when the environmental impact of a single output is investigated (Brankatschk & Finkbeiner, 2015; Goglio *et al*., 2018a). The lack of multifunctionality focus in agricultural LCAs has been suggested to favour intensified agricultural systems (van der Werf *et al*., 2020). By including some of these aspects, the investigation in Paper III revealed that the more extensive mixed grass-legume rotation led to the lowest GHG emissions per cereal unit. However, since the cereal unit is based on the animal feeding value of the crop, the conversion factor for the grass grown in the rotation was relatively high, although considerably lower than for the other crops in the rotation. This may be considered a drawback of using this method because the assessment is based on the agricultural outputs being used as feed, although there are other potential areas of use, such as food and bioenergy. However, in Paper III, this was believed to be a reasonable approximation since, in Sweden, most of the grass cultivated is used as fodder and cereals are largely used for animal feed rather than for human consumption (Eklöf, 2014; Cederberg & Henriksson, 2020). A similar approach is to assess "*the number of people actually fed per ha*" (Cassidy *et al*., 2013), expressed in edible energy and protein produced per ha and applied *e.g.* to assess sustainable and resilient farming strategies

(Röös *et al*., 2021). In this case, agricultural production is converted into a food value, which would serve as a fitting complement to the cereal unit.

In Paper IV, the effect of cover crop cultivation on subsequent crops was considered by estimating the nitrogen balance for each scenario assessed, in order to determine whether the cover cropping system resulted in an increase or decrease in demand for nitrogen fertiliser. This effect had a considerable influence on the overall climate effect of the system. However, the actual yield effect on the subsequent crop was not included in the analysis, due to lack of data. These type of linkages should be further examined in LCA, on both system and product level.

6.2.4 Life cycle assessment in policy and in research

Acceptance of the LCA methodology has increased over the years (Hellweg *et al*., 2023) and it is now recognised as a useful tool for policymakers and private organisations to assess the environmental impacts of products, processes and policy scenarios (Brandão *et al*., 2022b). However, the results are highly influenced by methodological choices. This means that the results of an LCA may not be directly comparable to results from another LCA, even if both studies are aligned with the international standard described in ISO (2006a) and ISO (2006b). This has raised doubts regarding the applicability of LCA in marketing or for constructing policy instruments, such as taxes, tariffs or procurement regulations, where it is crucial that the results are comparable. To overcome these issues, LCA frameworks such as PEF, EPD and RED have been developed (Brandão *et al*., 2022a). In these frameworks, it is important that the drawbacks of agricultural LCA mentioned above are acknowledged and considered, *e.g.* by including soil processes, the effect on crops grown on the same field, and spatial and temporal variability. Moreover, there is still a lack of some crucial indicators to describe the environmental impact of agricultural systems satisfactorily, such as land degradation, biodiversity loss and pesticide effects (van der Werf *et al*., 2020; Fan *et al*., 2022). Method development and consensus on how these impacts should be quantified are important for further utilisation of the method.

The ongoing work to make LCA results more comparable may have implications for use of the methodology in research, as the introduction of rigid rules can impose limitations, making it more difficult to adapt the methodology to answer specific questions. In research, the demand for comparability between LCA studies is not as crucial, since the aim and audience often differ. In addition, more effort is devoted to describing and justifying methodological choices. New insights from research should be incorporated into the frameworks when relevant, thereby driving the methodology forward, to better understand the environmental impacts of modern society and how they can be reduced.

7. Conclusions

Exploiting unused potential in the agricultural landscape is a promising measure to increase domestic biogas production and mitigate climate change.

- Introduction of a grass-based biogas system using fallow land in Uppsala Municipality resulted in a twofold increase in biogas production in the region, and led to mean climate change mitigation potential of 9950 tonnes CO_2e per year.
- Prioritising the best-performing sites in the region from a climate impact perspective led to lower climate impact per MJ biogas produced, but the greatest overall mitigation effect was achieved when all available land was employed.
- Cultivation of cover crops gave greater mitigation potential when the cover crop was harvested and utilised as feedstock in biogas production. The results showed high sensitivity to the timing of cover crop establishment, where earlier establishment led to higher yield and, consequently, a larger substitution effect. However, earlier establishment may affect cultivation of the preceding crop.
- Harvesting belowground cover crop biomass, in addition to aboveground biomass, increased the mitigation potential. However, this approach may introduce challenges related to biomass handling and conversion to biogas, as belowground biomass is associated with sand and grit contamination.
- The substitution effect of bioenergy is associated with uncertainties, as the displaced product may shift over time and in different contexts.

GHG emissions from cultivation show considerable spatial and temporal variation, which can have a significant influence on LCA results.

- GHG emissions within the systems studied exhibited large spatial and temporal variation. In fact, soil properties and weather conditions had a greater influence on the outcome than fertilisation rate for the simulated climate impact of grass cultivation, where the climate impact was greatest for heavy clay and carbon-rich soils, but lower for coarser soils with lower initial SOC stock. In addition, the climate impact of the grass cultivation systems increased over time, mostly due to decreased SOC sequestration over time. This highlights the importance of including spatial and temporal aspects of agricultural systems in order to improve the accuracy and reduce the uncertainty in LCA results.
- Soil N_2O emissions and changes in SOC stock resulted in considerable differences between sites. Hence, these processes are of high importance for site-dependent modelling aiming to include spatial variability in agricultural LCAs.
- Harvesting the cover crop resulted in a reduced potential for SOC sequestration. However, the modelling of the soil carbon fluxes showed that this reduction could be largely compensated if the biogas digestate was returned to the field and used as organic fertiliser.

Time- and space-dependent LCA increases data demand, but data gaps can be filled using process-based agricultural modelling.

- The increased data demand associated with time- and space-dependent life cycle inventories can be met using agricultural modelling to simulate inventory data. In this thesis, LCA was combined with the agro-ecosystem model DNDC to simulate grass cultivation under different conditions. This combined method could be used to design biomass production schemes in other regions, thereby serving as a strategic tool to assist land use planning of local energy production on arable land. The process-based models are, however, limited by scientific understanding of the described processes.
- The more simplistic soil carbon model ICBM was combined with the IPCC Tier I method to estimate soil N_2O emissions using system-

specific emissions factors. This approach is more transparent and straightforward, but is limited in that different processes such as crop growth, soil carbon and nitrogen fluxes are not included in the same model.

- Comparison of site-dependent methods used to estimate direct N_2O emissions revealed quite large difference in results, meaning that it was not possible to make any general recommendation on the best model to use.

Cultivation of grass and cover crops leads to higher soil organic carbon stock and offers an opportunity for atmospheric carbon dioxide removal and climate change mitigation.

- In the case studies in this thesis, the net effect of grass and cover crop cultivation was more carbon storage in the soil compared with the business-as-usual reference scenario, leading to a net reduction in GHG emissions from the agricultural system. The gross effect of the grass cultivation in some instances was loss of SOC over time, but at a slower rate than in the scenario without grass cultivation. Integrating two-year grasses (either pure grass or grass-legume mixture) in a six-year crop rotation was not sufficient to increase gross SOC stock, possibly due to high initial SOC content as a result of historical land use.
- Soil organic carbon sequestration potential was higher for soils with lower initial SOC content and high clay content. Increased carbon stock was achieved with a higher nitrogen fertiliser rate, which in turn led to elevated emissions of soil N_2O and emissions from fertiliser production, which offset the mitigation potential from sequestering carbon in the soil.
- Soil organic carbon sequestration does not provide permanent storage, which means that the stored carbon may be re-emitted to the atmosphere, *e.g.* due to changes in management practices or the environment. Hence, SOC sequestration cannot be used to offset CO_2 emissions from fossil sources, which cause climate forcing over thousands of years.

Grass and cover crop cultivation can be used to increase crop diversification, bringing benefits to the cropping system and reducing the demand for purchased agro-commodities, such as synthetic fertilisers.

- Re-cultivating unused land may increase resource demand, but diversification of cropping systems by including grass and cover crops in rotations may increase resource efficiency. In this thesis, inclusion of grass in crop rotations increased yield of the other crops in the rotation under a lower nitrogen fertilisation rate. This was especially evident on combining grass with nitrogen-fixing species (legumes).
- Cover crop cultivation was associated with reduced nitrogen leaching, which may decrease demand for synthetic fertilisers, reducing the climate impact of the system. However, the cover crop management practices that resulted in the highest SOC sequestration resulted in nitrogen being incorporated into soil organic matter, which led to a net increase in demand for nitrogen fertiliser.

8. Future research

Future research should continue to refine methodologies for effectively assessing the environmental impacts of agricultural and bioenergy systems. The areas suggested below offer avenues for expanding and enhancing understanding of grass and cover crop cultivation for biogas production and climate change mitigation.

Advancing soil process representation
Future research should focus on refining the representation of critical soil processes in relation to the climate impact of agricultural systems. The models used in this thesis are based on the current scientific understanding of these processes and there are still many knowledge gaps that need to be filled to improve the models. To reduce the inherent uncertainty of these models, more basic research is needed, particularly into the complex mechanisms underlying soil N_2O emissions and soil carbon fluxes and their interactions. Field trials involving continuous monitoring are also essential to better understand the spatial and temporal dynamics of these processes. Of particular relevance to the climate impact of biogas systems is the SOC effect following digestate application, for which reliable data are currently lacking.

Exploring regional and temporal variability
The findings in this thesis underscore the significant influence of spatial and temporal variability on the climate impact of cropping systems. Future investigations should delve deeper into the variation in climate impacts of crop cultivation, both between and within different regions. Innovative methodologies to include variation without heavily increasing data demand are urgently needed for agricultural LCA. Concurrent establishment of a comprehensive LCA database on regional heterogeneity would improve the accuracy of LCA and enhance its applicability as a decision-support tool.

Investigating alternative biomass utilisation

This thesis focused on utilisation of the biomass from grass cultivation and cover crops for biogas production. Future studies should explore use of the harvested biomass in more novel biorefinery platforms. One promising avenue is production of protein concentrates through biorefineries for high-quality feed or food applications.

Systemic effects of crop interactions

Future research should address the interactions and linkages between crops in cropping systems, in particular how the introduction of nitrogen-fixing leguminous crops affects the overall environmental impact of cropping systems, as it showed promise in results in reducing climate impacts in this thesis. Future research should also examine how crop rotation design to promote ecosystem services can reduce the demand for agronomic inputs to increase resource efficiency and improve domestic food and energy security, and assess how these crops can be integrated into food and bioenergy systems.

Policy and economy in a changing geopolitical landscape:

The effectiveness of grass- and cover crop-based biogas production systems in mitigating climate change rests not just on the environmental performance of the systems themselves, but also on the actual context of policy incentives and economic feasibility. Analysing the evolving roles of policies, subsidies, and market dynamics in shaping the scalability and adoption of these systems is critical, given the changing geopolitical landscape.

Improving LCA methodology:

While LCA is becoming more frequently used in policymaking, both in private and public organisations, is has some drawbacks that should be addressed in future research. These primarily concern assessment methodologies to include key environmental impacts in the agricultural sector such as biodiversity losses, land degradation and pesticide effects. Furthermore, the assessment of agricultural systems should be broadened beyond biomass provisioning to encompass various ecosystem services and deliverables.

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Popular science summary

One critical measure to mitigate climate change and increase energy security is to replace fossil fuels with renewable bio-based alternatives. One such renewable alternative is biogas produced through the anaerobic digestion of organic material. Biogas can be upgraded to biomethane, which can be used directly in current fossil fuel infrastructure, *e.g.* to produce heat and electricity, or as vehicle fuel. In addition to biogas, the anaerobic digestion produces digestate — an organic fertiliser that can be used to replace synthetic fertilisers.

One strategy to increase biogas production is to utilise energy crops, such as grass and cover crops[3]. Agricultural land is a limited resource, but in Sweden it is possible to increase biomass production *e.g.* by cultivation on previously unused agricultural land or increasing production in existing cultivation systems.

In the transition away from fossil fuels, it is crucial to study the emissions from bio-based alternatives, since studies have shown that some bio-based fuels can have significant climate impacts.

When evaluating the climate impact of crop-based bioenergy systems, it is important to consider emissions originating from cultivation of the biomass feedstock. The climate impact of energy crop cultivation is generated by agronomic inputs, such as fertilisers, agro-chemicals, machinery and energy. The efficiency of the system, i.e. how much output it produces in relation to input, will also affect the climate impact. Moreover, in agricultural systems, emissions arise from biological processes, such as changes in soil organic carbon stock and soil nitrous oxide (N_2O) and methane (CH_4) emissions. To complicate matters further, these emissions are

[3] Here “grass” refers to perennial grasses and legumes grown in pure or mixed stands and “cover crop” refers to crops grown between main crops in a rotation.

determined by *.e.g.* climate, management practices and soil properties, all of which vary over time and space.

Life Cycle Assessment (LCA) is a methodology that aims to include all environmental impacts throughout the life cycle of a studied product or system. However, variations in emissions over space and time and emissions from soil processes are often poorly represented in LCAs.

In this thesis, I combined LCA methodology with agricultural modelling of crop growth and soil carbon and nitrogen fluxes, and with data from short and long-term field experiments, to assess the climate mitigation potential of grass and cover crop cultivation and integration of these crops into biogas production systems in Sweden.

The results showed that grass and cover crop-based biogas systems had considerable climate change mitigation potential, via their ability to 1) increase resource use efficiency, 2) increase soil organic carbon stock and 3) produce bioenergy to replace fossil fuels. Inclusion of grass in crop rotation showed potential to increase the crop yields of the other crops in the rotation. This effect was especially evident when the grass was cultivated in a mixture including leguminous crops with the ability to fix nitrogen from the atmosphere. In addition, the grass and cover crop cultivation led to an increased soil carbon stock, which reduced the climate impact from the system. Increased nitrogen fertilisation rate resulted in a larger soil carbon stock, but also increased the GHG emissions from fertiliser production and soil borne N_2O emissions, which offset the mitigation potential of the soil carbon sequestration.

Simulations involving introduction of a grass-based biogas system in Uppsala Municipality using fallow land within the region showed potential to double biogas production and considerably reduce the Municipality's climate impact. However, the climate impact reduction of the biogas produced showed large variation, ranging between 102% and 79% compared with diesel fuel, depending on where in the Municipality the grass was cultivated. This variation was due to different soil properties and differences in distance to the biogas plant. Cultivation of oilseed radish as a cover crop was shown to have greater climate change mitigation potential when the crop biomass was harvested, primarily through fossil fuel substitution and a reduced risk of elevated nitrous oxide emissions associated with leaving the crop biomass in the field over winter. Larger climate change mitigation potential was assessed in the scenario where the taproot of the oilseed radish cover crop was harvested in addition to the aboveground biomass. The larger mitigation potential in this scenario was attributed to a larger biomass yield and biogas production, hence resulting in more diesel being replaced. Furthermore, the mitigation potential showed high sensitivity to the timing of cover crop establishment, where an early establishment led to larger yields. However, a too early establishment will affect the cultivation of the preceding crop rotation and is, therefore, limited to certain types of cropping systems.

The results of this thesis provide valuable insights that can be leveraged to develop sustainable crop-based bioenergy systems in Sweden as well as other regions with similar conditions.

Populärvetenskaplig sammanfattning

En viktig strategi för att begränsa den globala uppvärmningen och öka energisäkerheten är att ersätta fossila bränslen med förnybara biobaserade alternativ. Ett sådant förnybart alternativ är biogas som framställs genom rötning av organiskt material. Biogasen kan uppgraderas till biometan och användas direkt i befintlig fossil infrastruktur, exempelvis för att producera värme och el eller som fordonsbränsle. Vid sidan av biogasen produceras även i rötningsprocessen en rötrest som kan användas för att ersätta konstgödsel vars produktion är förknippad med stor klimatpåverkan.

Ett sätt att skapa förutsättningar för ökad biogasproduktion är att odla mer energigrödor, som vall och mellangrödor[4]. Jordbruksmark är en begränsad resurs som ska ge flera olika samhällsnyttor, men i Sverige finns potential att öka odlingen av vall och mellangrödor utan att konkurrera med annan befintlig markanvändning, exempelvis genom att använda tidigare outnyttjad jordbruksmark eller genom att öka produktionen i nuvarande odlingssystem.

I övergången från fossila bränslen är det avgörande att studera klimatpåverkan från de biobaserade alternativen, eftersom studier har visat att vissa biobaserade bränslen kan ha betydande klimatpåverkan.

Vid utvärdering av klimatpåverkan från grödobaserad bioenergi är det viktigt att ta hänsyn till utsläpp från odlingen av biomassaråvaran. Klimatpåverkan från odling av energigrödor genereras av agronomiska insatsvaror, såsom gödselmedel, jordbrukskemikalier, maskiner och energi. Även systemets effektivitet, dvs. hur mycket output det producerar i förhållande till input, påverkar systemets totala klimatpåverkan. I jordbrukssystem sker även klimatpåverkan från biologiska processer, exempelvis genom förändring av markens kollager samt markbundna utsläpp

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[4] Vall är fleråriga gräs och baljväxter som odlas i rena eller blandade bestånd och mellangrödor är grödor som odlas mellan huvudgrödor i en växtföljd

av de potenta växthusgaserna N_2O och CH_4. För att komplicera saken ytterligare bestäms dessa utsläpp bland annat av klimat, skötselmetoder och markens egenskaper, vilka varierar mellan olika platser och över tid.

Livscykelanalys (LCA) är en metod som syftar till att inkludera all miljöpåverkan under hela livscykeln för en studerad produkt eller ett studerat system. LCA-studier inkluderar dock sällan variationer i utsläpp över tid och rum samt utsläpp från markprocesser.

I den här avhandlingen kombinerades LCA-metodik dels med jordbruksmodellering av grödans tillväxt samt kol- och kväveflöden i marken, dels med data från fältförsök för att bedöma klimatpåverkan från odlingen av vall och mellangrödor och deras integrering i biogasproduktionssystem i Sverige.

Resultaten visade att odling av vall och mellangrödor för biogasproduktion har potential att generera betydande utsläppsminskningar, genom att 1) öka effektiviteten i resursanvändningen, 2) öka markens kollager samt 3) producera bioenergi som kan användas för att ersätta fossila bränslen. Införandet av vall i växtföljder visades ha potential att öka skörden av de övriga grödorna i växtföljden. Denna effekt var särskilt tydlig när vallen innehöll kvävefixerande baljväxter. Dessutom ledde odlingen av vall och mellangrödor till ett ökat kollager i marken, vilket minskade systemets klimatpåverkan. En högre gödselgiva resulterade i en större kolinlagring, men ledde också till ökade utsläpp från produktionen av konstgödseln samt från markbundna lustgasemissioner, vilket motverkade utsläppsminskningen från markkolsinbindningen.

Simuleringar av införandet av ett vallbaserat biogassystem i Uppsala kommun visade potential att fördubbla biogasproduktionen och markant minska kommunens klimatpåverkan. Utsläppsminskningen från den producerade biogasen varierade dock kraftigt, mellan 102 % och 79 % jämfört med diesel, beroende på var i kommunen vallen odlades. Denna variation berodde på olika markegenskaper och skillnader i avstånd till biogasanläggningen. Odling av oljerättika som mellangröda visade sig ha större potential till utsläppsminskning när grödans biomassa skördades, främst genom att ersätta fossila bränslen, men även genom att minska risken för förhöjda lustgasutsläpp i samband med att grödan lämnas kvar i fält över vintern. Större utsläppsminskning bedömdes för ett scenario när även oljerättikans pålrot skördades, då detta gav en större skörd och därmed mer biogas, vilket i sin tur ledde till att mer fossil diesel kunde ersättas.

Potentialen för utsläppsminskningen var starkt beroende av etableringstidpunkt för mellangrödan, där en tidigare etablering i regel resulterade i en högre skörd. En för tidig etablering riskerar dock påverka den föregående grödan och är därför begränsad till vissa typer av odlingssystem.

Dessa resultat ger värdefulla insikter som kan användas för att skapa hållbara grödobaserade bioenergisystem i Sverige och i andra regioner med liknande förhållanden.

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Assessing the climate and eutrophication impacts of grass cultivation at five sites in Sweden

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ABSTRACT
In this study, Life Cycle Assessment (LCA) methodology was combined with the agro-ecosystem model DNDC to assess the climate and eutrophication impacts of perennial grass cultivation at five different sites in Sweden. The system was evaluated for two fertilisation rates, 140 and 200 kg N ha^{-1}. The climate impact showed large variation between the investigated sites. The largest contribution to the climate impact was through soil N_2O emissions and emissions associated with mineral fertiliser manufacturing. The highest climate impact was predicted for the site with the highest clay and initial organic carbon content, while lower impacts were predicted for the sandy loam soils, due to low N_2O emissions, and for the silty clay loam, due to high carbon sequestration rate. The highest eutrophication potential was estimated for the sandy loam soils, while the sites with finer soil texture had lower eutrophication potential. According to the results, soil properties and weather conditions were more important than fertilisation rate for the climate impact of the system assessed. It was concluded that agro-ecosystem models can add a spatial and temporal dimension to environmental impact assessment in agricultural LCA studies. The results could be used to assist policymakers in optimising use of agricultural land.

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Carbon sequestration; DNDC model; greenhouse gas emissions; life cycle assessment (LCA); perennial cropping systems; soil N_2O emissions

Introduction

Perennial grasses are one of the most commonly grown crops in humid and cold regions. They are primarily grown for forage in animal husbandry, but other alternative uses such as feedstock for bioenergy production have been proposed (Tilman et al. 2006; Auburger et al. 2017; Carlsson et al. 2017). Earlier studies have shown that soil organic carbon (SOC) is often more abundant in perennial than in annual cropping systems, an effect attributed to increased carbon (C) inputs due to high root biomass turnover, less exposure to ploughing and a longer growing season compared with annual crops (Baker et al. 2007; Bolinder et al. 2010; Börjesson et al. 2018). This feature is interesting from a global warming mitigation perspective (Smith et al. 2016; Minx et al. 2018) and soil C sequestration through grass cultivation has been suggested as a negative emission technology with large potential (Tidåker et al. 2014; Yang et al. 2018). However, grass-ley systems have been reported to act differently depending on climate and soil properties (Soussana et al. 2010; Kätterer et al. 2012; Jackson et al. 2017).

Grass cultivation also inevitably has environmental impacts due to different inputs during the system lifecycle, and it is important to determine these impacts in order to assess the full environmental burden of the system. For example, pure grass swards are reliant on inputs of fertilisers to promote high biomass yield and achieve high soil C sequestration (Yang et al. 2018). Mineral fertiliser use in agriculture is associated with environmental impacts, primarily global warming and eutrophication. The climate impact of mineral fertiliser is caused by both manufacturing and soil application, the latter by inducing increased terrestrial emissions of the greenhouse gas (GHG) nitrous oxide (N_2O). This GHG is of particular importance since it contributes significantly to the climate impact (Bouwman et al. 2002). Estimates of N_2O emissions are associated with

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considerable uncertainty, due to substantial temporal and spatial deviations and because the underlying processes affecting emissions are still not fully known (Butterbach-Bahl et al. 2013). Management of the cultivation system, such as field operations, will also affect the total environmental impact (Tidåker et al. 2014).

Life Cycle Assessment (LCA) is a comprehensive approach for investigating the environmental impact of products and services. The method was originally developed as a site-independent tool for industrial processes, but has also been widely used for assessment of the environmental impact of agricultural systems (Garrigues et al. 2012). In contrast to the impacts of most industrial processes, the environmental impacts of agriculture are determined by, and embedded in, physical, climatological, social and environmental conditions. Moreover, these determinants vary over time and space. This means that where and when the cultivation takes place will affect the environmental impact of the studied system, for example due to variations in climate and soil properties (Miller et al. 2006). These variations have been proven to be important (Humpenöder et al. 2013; Hörtenhuber et al. 2014; Henryson et al. 2019), but are rarely included in LCA studies, often because of the extensive data demand and since measurements of these processes are time-consuming and costly. Thus in LCA analyses most practitioners rely on databases with low temporal and spatial resolution (Rebitzer et al. 2004).

One approach to include the spatial and temporal variations of the life-cycle impact of agricultural systems is to combine LCA methodology with agro-ecosystem modelling (e.g. Bessou et al. 2013; Goglio et al. 2014; Kløverpris et al. 2016; Deng et al. 2017). The DNDC model is a well-recognised, process-based biogeochemical model that has been used for sites all over the world (Giltrap et al. 2010; Gilhespy et al. 2014; Brilli et al. 2017; Ehrhardt et al. 2018). Since the first version was launched, developers have successively improved the model with additional agro-ecosystem mechanisms (Gilhespy et al. 2014). The DNDC model has been used for example to fill data gaps in LCAs in recent studies (Goglio et al. 2014, 2018).

In this study, we assessed the potential climate impact and eutrophication potential of grass cultivation at five sites in Sweden with different characteristics. The DNDC model was used to simulate C and N fluxes and calculate site-dependent impacts, in a life cycle perspective. The system boundary was set from cradle to farm gate, and the environmental impact was calculated per hectare and per Mg dry matter (DM) yield. Since estimates of N_2O emissions from soil sources have a high degree of uncertainty, we opted to compare three methods for calculating these emissions.

Material and methods

Site-specific data for each of the five sites were used to model life cycle inventory data, which were then used to evaluate the environmental impact of the grass cultivation system. The inventory data collected to assess the climate impact of the system comprised field operations (including sowing, rolling, cutting and ploughing), manufacturing of mineral N fertiliser and soil N_2O (direct and indirect), CH_4 and C fluxes, with the latter three estimated using the DNDC model. For the eutrophication assessment, the life cycle inventory was conducted using nitrogen (N) and phosphorus (P) leaching data from Johnsson et al. (2016).

Experimental sites

The five study sites selected were distributed over southern and central Sweden (Figure 1), to cover variations in climate and soil properties. The soils at the two most northerly sites, Kungsängen (59.8°N) and Karlslund (59.4°N), both had a high clay content (57% and 29%, respectively) and initial SOC content (6.0% and 2.6%, respectively). The soil at Lanna (58.5°N) was a silty clay loam with lower SOC content (2.0%) than the two soils at higher latitudes and

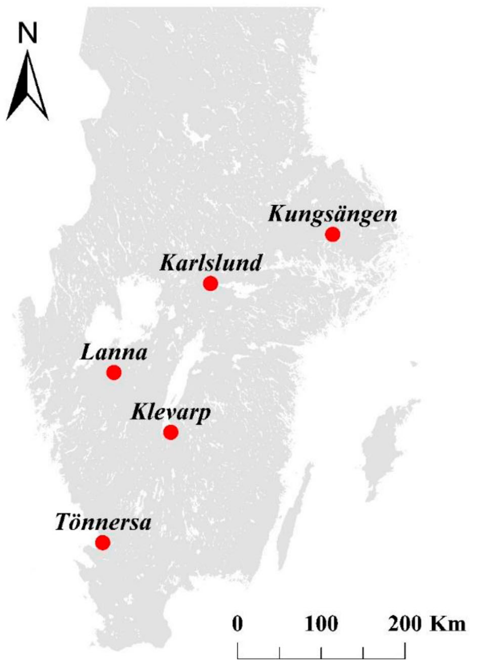

Figure 1. Map of southern and central Sweden indicating the location of the five study sites.

Table 1. Specific characteristics of the five study sites.

Site	Karlslund	Klevarp	Kungsängen	Lanna	Tönnersa	Source
Latitude	59.4	57.7	59.8	58.5	56.5	Eckersten et al. (2004)
Mean temp (°C) 1986–2015	6.8	5.4	6.9	7.1	8.0	SMHI
Mean annual precipitation (mm) 1986–2015	691	679	568	598	791	SMHI
N in precipitation (ppm)	1.2	1.5	1.2	1.4	1.7	Krondroppsnätet (2018)
Soil texture	Clay loam	Sandy loam	Clay	Silty clay loam	Sandy loam	Eckersten et al. (2004)
Soil organic carbon at surface (%)	2.6	1.7	6.0	2.0	1.5	Eckersten et al. (2004)
Clay fraction (%)	29	2.1	57	33	7.2	Eckersten et al. (2004)
Sand fraction (%)	33	65	30	10	65	Assumption
Bulk density (g/cm^3)	1.29	1.37	1.39	1.24	1.43	Estimation based on Saxton and Rawls (2006)
Porosity (%)	51	48	48	53	46	Estimation based on Saxton and Rawls (2006)
Field capacity (water-filled pore space)	0.67	0.31	0.87	0.72	0.36	Estimation based on Saxton and Rawls (2006)
Wilting point (water-filled pore space)	0.38	0.09	0.71	0.39	0.14	Estimation based on Saxton and Rawls (2006)

Note: Data on nitrogen (N) concentration in precipitation were obtained from the national inventory database (Krondroppsnätet 2018). No values for the period of interest were available for the Kungsängen site and therefore the concentration for Karlslund, the nearest site to Kungsängen, was used. The sand fraction was assumed based on average soil texture values. SMHI: https://www.smhi.se/klimatdata & Krondroppsnätet: http://krondroppsnatet.ivl.se.

with 33% clay content. The two most southerly sites, Klevarp (57.7°N) and Tönnersa (56.5°N), both had sandy loam soils with low SOC content (1.7% and 1.5%, respectively). Tönnersa had the highest mean annual temperature and precipitation of all five sites and Klevarp, located in the centre of the south Swedish highlands, had the lowest mean annual temperature. Soil and climate properties for each site are shown in (Table 1).

Perennial cropping system

A five-year grass cultivation system was simulated over 30 years for each of the individual sites, using weather data for the period 1986–2015 (Table S1 in Supplementary Material (SM)). Each rotation started with sowing and rolling in the first year and ended with ploughing to 30 cm depth in year five (Figure 2). During the crop rotation, the grass was fertilised with mineral N fertiliser and cut twice a year. Two fertilisation rates were compared, F1 = 140 kg N ha^{-1} and F2 = 200 kg N ha^{-1}. Spreading of fertiliser was split between two occasions each year, with the first application (80/120 kg N ha^{-1}) on 1 May and the second (60/80 kg N ha^{-1}) on 10 June, shortly after the first cut.

Modelling and assumptions

Agro-ecosystem modelling

The DNDC model is driven by climate, soil, vegetation and management variables, which are used to simulate critical terrestrial processes such as crop growth, soil C dynamics, soil temperature and moisture regimes and emissions of greenhouse and trace gases. The simulation results are dynamically presented on a daily time step (Li et al. 1992, 2012). In this study, we used a model version that contains more detailed descriptions of crop biomass growth (Kröbel et al. 2011), soil temperature (Dutta et al. 2017) and evapotranspiration (Dutta et al. 2016), and has recently been modified for simulating perennial regrowth after each cut and in subsequent years (He et al. 2019). This version was chosen because it has been used to simulate perennial growth in similar cool-weather conditions to those in Sweden (He et al. 2019). The model was used to estimate life cycle inventory data for soil C fluxes, N_2O and CH_4 emissions and biomass yield, assuming that 85% of aboveground biomass was harvested at every cut. The parameterisation of the model is presented in (Table S2) in SM. Indirect N_2O emissions were calculated using the default emission factor (0.0075) from IPCC (2006) associated with N leaching and runoff, which were simulated using the DNDC model.

Field trials designed to study the growth pattern of a mixture of timothy grass (*Phleum pratense* L.) and meadow fescue (*Festuca pratensis* Huds.) over two consecutive years were conducted at the study sites between 1985 and 1988. At Kungsängen and Klevarp, the two-year trials were performed twice, i.e. for four years in total. All fields were treated equally in order to make the results comparable. For more information about the experimental set-up, see Eckersten et al. (2004, 2007). The DNDC model was evaluated for simulating the biomass growth pattern over the growing seasons. Data displayed in (Table 1) were used as input in the model to define the conditions at the different sites. Root:shoot ratio was assumed to be 1, i.e. 50% of total biomass, based on previous grass cultivation modelling studies by Eckersten et al. (2004) and Johnsson

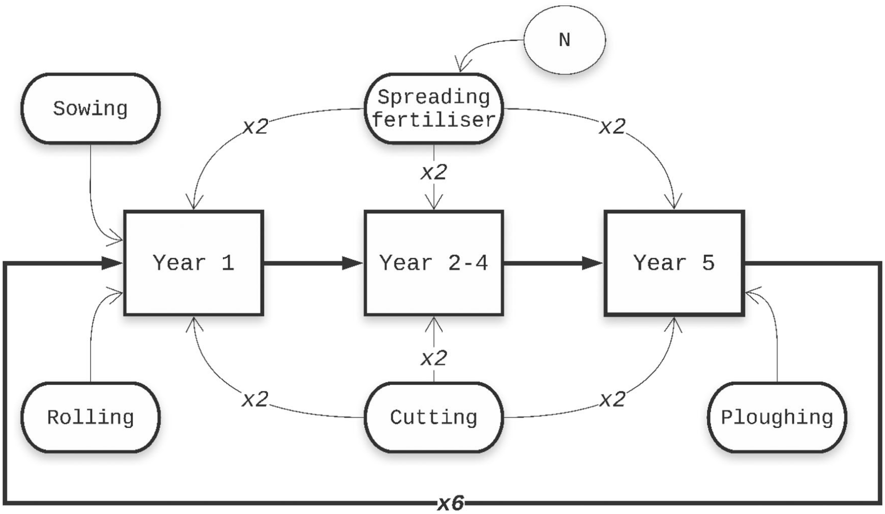

Figure 2. Overview of the crop rotation simulated for all study sites. The grass was sown and the soil was rolled in the first year, then growth continued for five more years. During this period, fertiliser was applied and the grass was cut twice every year. The crop rotation ended with deep ploughing to 30 cm. The rotation was repeated six times.

et al. (2016). The model fit to observed growth data was evaluated as coefficient of determination (r^2, eq. 1), normalised root mean square error (nRMSE, eq. 2) and Nash-Sutcliffe model efficiency coefficient (Nash and Sutcliffe 1970) (ME, eq. 3). An ME value > 0 corresponds to goodness of fit better than the observed mean value, while ME = 1 corresponds to a perfect fit. To evaluate model performance, the goodness of fit statistics were calculated for all biomass data and the biomass observations closest to harvest.

$$r^2 = \left(\sum_{t=1}^{n} (S_t - \bar{S})(O_t - \bar{O}) \right)^2 / \sum_{t=1}^{n} (S_t - \bar{S})^2 \sum_{t=1}^{n} (O_t - \bar{O})^2 \tag{1}$$

$$n\text{RMSE} = \left(\sum_{t=1}^{n} (O_t - S_t)^2 / n \right)^{1/2} / \bar{O} \tag{2}$$

$$\text{ME} = 1 - \sum_{t=1}^{n} (O_t - S_t)^2 / \sum_{t=1}^{n} (O_t - \bar{O})^2 \tag{3}$$

where O denotes observed biomass and S simulated biomass.

Soil N_2O method comparison

Earlier studies have shown the importance of N_2O emissions when examining the climate impact of agro-ecosystems (e.g. Jury et al. 2010; Ruan et al. 2016). Because of the uncertainties associated with estimating soil-borne N_2O, we compared the results from the DNDC model with those obtained using two empirical approaches. These were: (i) the IPCC Tier 1 site-generic emissions factor, 0.01 kg N_2O-N kg N^{-1}, assuming no change in soil C stocks (IPCC 2006), and (ii) a site-specific approach developed by Rochette et al. (2018) who concluded, based on N_2O emissions observations in Canada, that cumulative emissions from synthetic N application, N_2O_{Roch} (kg N_2O-N ha^{-1}), can be predicted successfully ($R^2 = 0.68$) with the equation:

$$N2O_{Roch} = \exp(3.91 + 0.0022P + 0.0069\text{Min}N - 0.0032S\text{AND} - 0.747pH + 0.097T_{air}) \tag{4}$$

where P is growing season precipitation (mm), *MinN* is mineral N application (kg), *SAND* is soil sand content (g kg^{-1}), *pH* is soil pH and T_{air} is mean annual air temperature (°C) (Rochette et al. 2018).

The three methods were compared by calculating the yearly cumulative direct N_2O emissions at each of the five study sites.

Field operations and fertiliser manufacture

Diesel consumption for sowing, rolling and spreading fertiliser was assumed to be 2.3, 2.3 and 4.7 L ha^{-1}, respectively (Carlsson et al. 2017). Diesel consumption for cutting and ploughing was based on linear regression models with biomass yield and clay content, respectively, as the independent variable (Arvidsson and Keller 2011; Prade et al. 2015). The GHG emissions from production and use of diesel were set to 2.8 kg CO_2, 1.2 g CH_4 and 0.073 g N_2O L^{-1}, based on Gode et al. (2011). The GHG emissions during manufacture of mineral fertiliser were set to 3.5 kg CO_2-eq kg^{-1} N, where the climate impact was assumed to be 86% from CO_2 emissions, with the remaining 14% from N_2O (Bentrup et al. 2016).

Nitrogen and phosphorus leaching

Nitrogen and *P* leaching were estimated using data from Johnsson et al. (2016), who performed national simulations of mean leaching rates in 22 different regions in Sweden, using the models SOILNDB for N and ICECREAMDB for P leaching. The data represent leaching from the root zone and surface runoff for specific crops and soil textures (Johnsson et al. 2016).

Climate impact assessment

The climate impact was assessed using Global Warming Potential (GWP) and dynamically using Absolute Global Temperature Potential (AGTP), as defined by the IPCC (Myhre et al. 2013). The GWP methodology compares the cumulative radiative forcing of a GHG emission with the radiative forcing of an equal amount of emitted CO_2 over a specific period, typically 100 years (Myhre et al. 2013). The characterisation factor for CH_4 and N_2O is 34 and 298, respectively, with the inclusion of climate-carbon feedbacks (Myhre et al. 2013). One of the limitations with the GWP approach is that the method does not include the timing of emissions, which means that emissions which occur during different points in the life cycle are added together, although the endpoint of the impact differs (Kendall 2012).

The AGTP approach goes one step further by analysing the potential temperature change due to the change in radiative forcing caused by a pulse emission of GHGs, which is achieved by applying radiative forcing calculations in convolution with the climate temperature response to changes in radiative forcing. By investigating the cumulative temperature response from the yearly emissions modelled in the life cycle inventory, the climate impact can be assessed dynamically throughout a specified analytical time horizon. This approach to assessing the climate impact has been used previously in LCA studies to evaluate the climate impact of bioenergy systems (Ericsson et al. 2013; Hammar et al. 2017).

Table 2. Marine eutrophication and potential eutrophication at the study sites, calculated using nitrogen (N) and phosphorus (P) characterisation factors (CF) taken from CML (Guinée, 2002) and from Henryson et al. (2018), respectively.

	Marine eutrophication Henryson *et al.* (kg N-eq kg^{-1})		Potential eutrophication CML (kg N-eq kg^{-1})	
Sites	N CF	*P* CF	N CF	*P* CF
Karlslund	0.169	0.672	1	7.23
Klevarp	0.122	0.499	1	7.23
Kungsängen	0.435	2.48	1	7.23
Lanna	0.55	0	1	7.23
Tönnersa	0.835	0	1	7.23

Eutrophication impact assessment

The eutrophication caused by the leached N and P was assessed using two different, but complementary methods. First of all, we used the site-generic CML methodology (Guinée, 2002) to assess the potential eutrophication impact of estimated N and P leaching. This method places the indicator at the point of emission and thus neglects the fate of the eutrophying emissions. Furthermore, the method considers all N and P discharged to the environment as having eutrophying capacity and includes all recipients, such as terrestrial, freshwater and marine water bodies (Guinée, 2002). In reality, eutrophication is more complicated and highly dependent on spatial properties. One example is the Baltic Sea, which is the world's largest brackish water basin and, unlike most marine environments, is considered limited by both N and P, with variations between different sub-basins (Swedish EPA 2006). To account for this, we used site-specific marine eutrophication characterisation factors developed by Henryson et al. (2018) for different regions in Sweden. These include site and catchment properties and the P or N limiting status of the recipient, and were used here as a complement to the CML calculations to investigate the impact on the complex marine environment that surrounds Sweden. The characterisation factors used in the CML and Henryson *et al*. approach are listed in (Table 2).

Results

Life cycle inventory

The climate impact inventory was divided into change in SOC content, soil N_2O and CH_4 emissions, fertiliser manufacturing and field operations. The results of the life cycle inventory for soil C balance and soil N_2O emissions are presented in section 3.1.1 and the results of the inventory analysis of eutrophying N and P leaching rates in section 3.1.2.

SOC balance and N_2O emissions

The soil at all sites investigated showed an ability to sequester C over the complete simulation period and for both fertilisation rates, although the increase was low (0.035 Mg ha^{-1} in treatment F1) for the site with initial highest SOC content (Kungsängen). The largest increase in SOC content was for the silty clay loam at Lanna (4.3 and 6.5 Mg ha^{-1} over the 30-year simulation period for F1 and F2, respectively). As expected, the F2 application rate led to greater C sequestration in all soils than the F1 rate (Figure 3). At the end of each crop rotation, all living biomass (aboveground and belowground) was terminated through ploughing and thereby transferred to the SOC pool, which explains the large SOC increase every fifth year in (Figure 3). Yearly mean gross C input, i.e. before degradation, for all soils, was 2.7 and 3.4 Mg C ha^{-1} for F1 and F2, respectively.

Simulated cumulative N_2O emissions were highest for the clay and SOC-rich soil in Kungsängen (mean 5.2 kg N_2O ha^{-1} y^{-1} for F1 and 6.1 kg N_2O ha^{-1} y^{-1} for F2), while emissions were lower for the sandy loam soils at Klevarp and Tönnersa. The Klevarp site had the lowest estimated emissions (1.9 kg N_2O ha^{-1} y^{-1} for F1 and 2.1 kg N_2O ha^{-1} y^{-1} for F2). Higher emissions from soils containing more clay are consistent with findings in a meta-analysis based on observations from Rochette et al. (2018). There was considerable variation between simulated years, especially for the Kungsängen soil (Figure 4). This annual variation was attributed to weather fluctuations, for example differences in amount and pattern of precipitation. Mean N_2O emissions over the simulation period were slightly higher for the higher fertilisation rate (F2) at all study sites.

The different methods to estimate N_2O emissions were compared by calculating the emissions for each site. The two site-specific methods, DNDC and Rochette *et al.*, showed large variation between the different sites. Overall, the DNDC model predicted higher annual emissions than the Rochette *et al.* approach (Figure S1 in SM). The DNDC model predicted the highest emissions rate for the clay-rich soil at Kungsängen, while the Rochette *et al.* approach predicted the highest emissions for the field at Lanna, with the lowest soil sand content. Both site-dependent methods predicted the lowest emissions from the sandy loam soils at Klevarp and Tönnersa. Mean emissions across all sites calculated with the DNDC, Rochette *et al.* and IPCC Tier 1 approaches were 1.97 ± 0.83, 1.41 ± 0.93 and 1.63 ± 0.02, respectively, for F1 and 2.29 ± 0.98, 2.13 ± 1.41 and 2.31 ± 0.02 kg N_2O-N ha^{-1}, respectively, for F2. Mean estimates for each field are shown in (Table 3).

Nitrogen and phosphorus leaching

Nitrogen leaching was estimated to be higher for the sandy loam soils at Tönnersa and Klevarp than for the soils with higher clay content at Kungsängen, Lanna and Karlslund. The lowest N leaching rate was predicted for the soil with the highest clay content (Kungsängen). For P leaching, the trend was roughly the opposite, i.e.

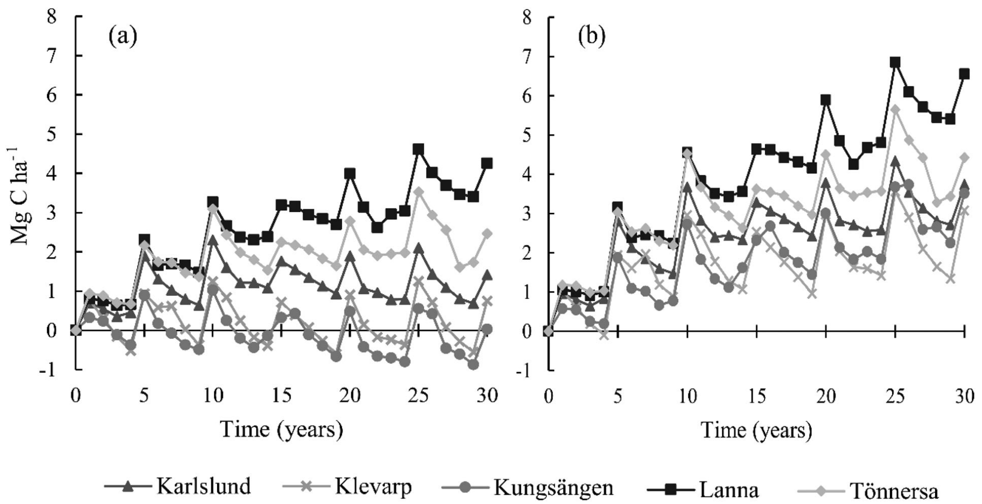

Figure 3. Simulated cumulative change in soil organic carbon (SOC) for fertiliser rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-}) over the 30-year study period. The SOC change is presented in Mg C ha^{-1}.

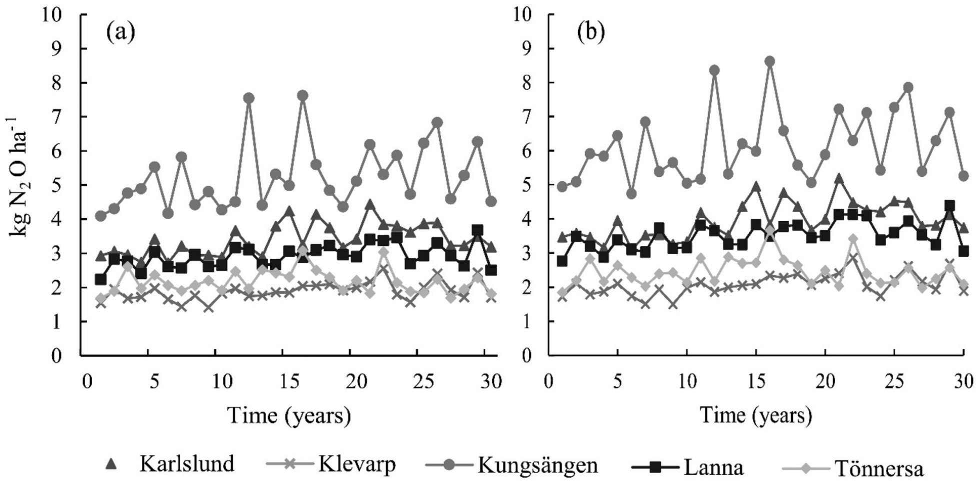

Figure 4. Annual cumulative nitrous oxide emissions from the study sites (kg N_2O ha^{-1}), estimated using the DNDC model, for fertilisation rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-1}) over the 30-year simulation period.

Table 3. Mean nitrous oxide (N_2O) emissions at the five study sites under fertilisation rates F1 and F2 (140 and 200 kg N ha^{-1}, respectively), assessed with three different approaches: DNDC, Rochette and IPCC Tier 1.

Sites	DNDC (kg N_2O-N ha^{-1})	Rochette (kg N_2O-N ha^{-1})	IPCC Tier 1 (kg N_2O-N ha^{-1})
Karlslund F1	2.15 ± 0.29	1.50 ± 0.29	1.63 ± 0.01
Klevarp F1	1.19 ± 0.17	0.47 ± 0.09	1.64 ± 0.01
Kungsängen F1	3.32 ± 0.59	1.41 ± 0.24	1.61 ± 0.02
Lanna F1	1.84 ± 0.21	2.98 ± 0.55	1.62 ± 0.01
Tönnersa F1	1.37 ± 0.22	0.69 ± 0.17	1.63 ± 0.01
Karlslund F2	2.49 ± 0.33	2.27 ± 0.43	2.31 ± 0.01
Klevarp F2	1.30 ± 0.20	0.71 ± 0.13	2.32 ± 0.02
Kungsängen F2	3.89 ± 0.64	2.13 ± 0.36	2.28 ± 0.03
Lanna F2	2.22 ± 0.25	4.50 ± 0.83	2.30 ± 0.02
Tönnersa F2	1.54 ± 0.26	1.05 ± 0.26	2.32 ± 0.02

with higher leaching for the clay-rich soils than the sandy soils. The highest P leaching was predicted for the soil with 33% clay content (Lanna) (Table 4).

Life cycle impact assessment

The results from the life cycle inventory were used to assess the climate impact and potential eutrophication

Table 4. Predicted mean nitrogen (N) and phosphorus (*P*) leaching for the five study sites.

Site	N (kg ha^{-1})	*P* (kg ha^{-1})
Karlslund	3	0.41
Klevarp	15	0.27
Kungsängen	1	0.43
Lanna	3	0.79
Tönnersa	18	0.23

impact of the grass cultivation system at each of the five study sites.

Climate impact

The GHG fluxes from the inventory analysis were divided into five categories and analysed with GWP_{100} (Figure 5). Mean total GHG emissions for all five sites were 1170 ± 460 and 1200 ± 460 kg CO_2-eq ha^{-1} y^{-1} for F1 and F2, respectively. Expressed per Mg DM, the mean emissions were 178 ± 77 and 136 ± 59 kg CO_2-eq for F1 and F2, based on the 30-year simulation. The large standard deviation indicates considerable variation between the sites. The highest emissions were simulated for Kungsängen (321 and 244 kg CO_2-eq Mg DM^{-1} for F1 and F2, respectively) and the lowest for Tönnersa (89 kg CO_2-eq Mg DM^{-1} for F2). The total climate impact of the system was mainly a balance between increased soil C stocks, i.e. C sequestration, and emissions of N_2O from soil processes and GHG emissions from manufacturing of the fertiliser. The grass cultivation resulted in a small CH_4 sink for all simulated sites (Figure 5). The higher fertilisation rate (F2) generated lower emissions per Mg DM in all fields, due to more soil C sequestration and higher grass yield. However, the variation between sites was greater than that between fertiliser rates.

The relationship between the emissions categories shown in (Figure 5) changed for different rotation periods over the 30-year simulation period. In other words, the climate impact assessed as GWP varied not only between sites and fertilisation rates, but also over time between consecutive rotations throughout the

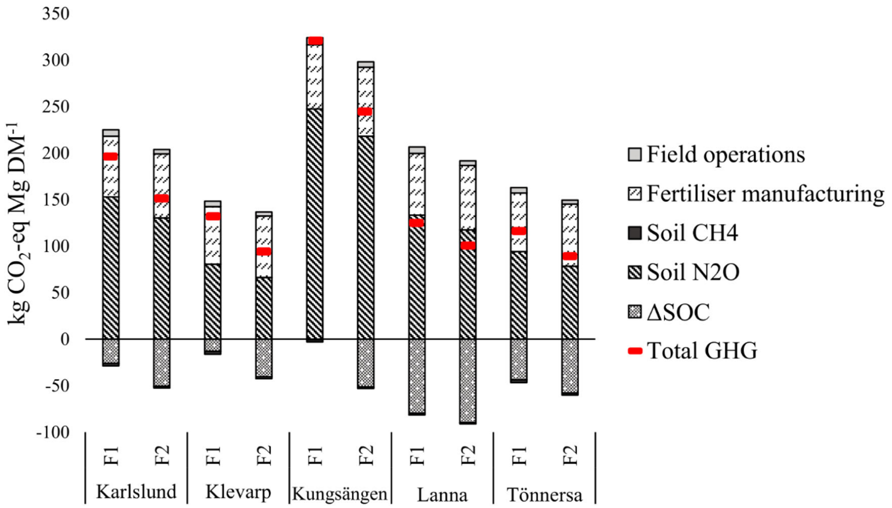

Figure 5. Total climate impact of grass cultivation during the 30-year simulation period at the five study sites for fertilisation rates F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}), assessed as Global Warming Potential over 100 years (GWP100). SOC = soil organic carbon, GHG = greenhouse gases.

study period. For all fields except Kungsängen (F1 and F2) and Klevarp (F1), the first cropping sequence demonstrated a global warming mitigating effect, whereas the last rotation enhanced the global warming effect at all sites (Table 5). The main reason for this was that the soil C sequestration rate was higher during the first rotation compared with the last.

The climate impact was further investigated using the dynamic climate impact assessment model described in section 2.4. The yearly GHG fluxes from the system were used to calculate the cumulative temperature change for 100 years, expressed as pK ha^{-1} ($p = 10^{-12}$). The change in global mean temperature due to grass cultivation at the study sites is shown in (Figure 6). Similarly to (Table 5), it shows a lower temperature change at the beginning of the simulation period and an increasing rate of impact over time. At the temporal boundary of the system, i.e. after 30 years, the climate impact increased for a few years before it started to decline, which was due to the atmospheric inertia related to GHG emissions and temperature increase. Seventy years beyond the system's temporal boundary, grass cultivation still had a warming effect on the climate.

Table 5. Climate impact assessed as Global Warming Potential (GWP) for the first crop rotation (1) and the last (6) at the different sites under fertilisation rate F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}).

Site and fertilisation rate	Crop rotation 1 (kg CO_2-eq Mg DM^{-1})	Crop rotation 6 (kg CO_2-eq Mg DM^{-1})
Karlslund F1	−4	290
Klevarp F1	39	206
Kungsängen F1	202	398
Lanna F1	−70	244
Tönnersa F1	−76	252
Karlslund F2	−42	245
Klevarp F2	−22	177
Kungsängen F2	119	322
Lanna F2	−85	215
Tönnersa F2	−99	227

Note: The results for each site are expressed per Mg DM.

Eutrophication assessment

The potential eutrophication (CML) and marine eutrophication (Henryson *et al.*) impact of N and P leaching were assessed on a per-hectare basis (Figure 7). Mean eutrophication potential for all sites, assessed with CML characterisation factors, was 11.1 ± 6.1 kg N-eq ha^{-1} (range 4.1 kg N-eq ha^{-1} for Kungsängen to 19.7 kg N-eq ha^{-1} for Tönnersa). The high eutrophication potential at Tönnersa was mainly due to the high N leaching rate at that site. In general, the eutrophication potential was higher for the sandy loam soils at Tönnersa and Klevarp and lower for the more clay-rich soils at the other sites.

Mean marine eutrophication at all sites, assessed with the Henryson *et al.* approach, was 4.2 ± 5.4 kg N-eq ha^{-1} (ranging from 0.1 kg N-eq ha^{-1} at Karlslund to 15.0 kg N-eq ha^{-1} at Tönnersa) (Figure 7). The lower impact compared with the CML approach is because the Henryson *et al.* characterisation factors assess marine

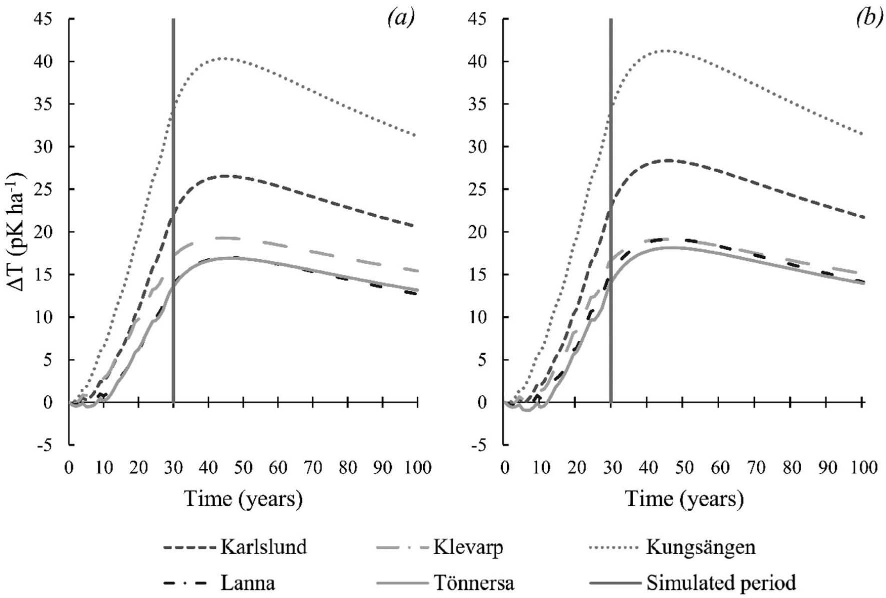

Figure 6. Simulated potential temperature response of the grass cultivation system during the 30-year study period with fertilisation rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-1}) at the five study sites. The temperature response is expressed as pK ha^{-1} ($p = 10^{-12}$, K = Kelvin).

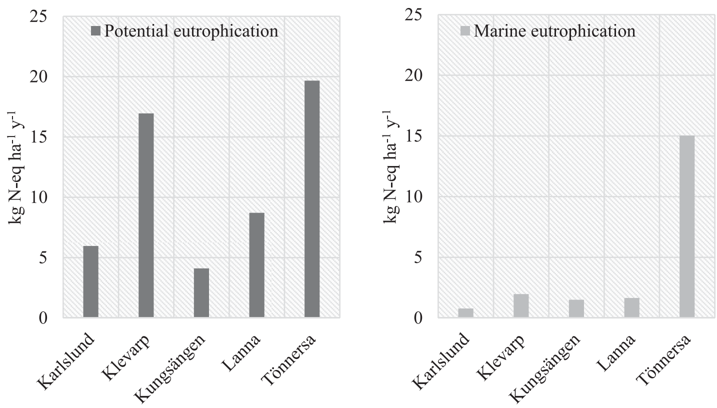

Figure 7. Potential eutrophication impact assessed using CML (potential eutrophication) and Henryson (marine eutrophication) methodology. The bar represents the eutrophication in kg N-eq ha^{-1} y^{-1}.

eutrophication exclusively, which means that results derived with the different methods should not be compared directly. However, it is relevant to analyse how the pattern of the estimated eutrophication differed between the two approaches. For instance, according to the Henryson *et al.* method, the eutrophication level for the sandy soil at Klevarp was similar to that for soils with a higher clay content. The other sandy loam soil (Tönnersa) was estimated to have the highest marine eutrophication, because of high N leaching rate and proximity to the recipient. Compared with the CML approach, relatively lower eutrophication was assessed for the field in Lanna, partly because of N-limiting characteristics of the recipient.

Model biomass growth goodness of fit

The goodness of fit of the DNDC model was analysed using mean simulated and observed aboveground biomass at each cutting occasion. This analysis showed that the mean simulated aboveground biomass was within the standard deviation of the observed data for each site (Figure 8). The goodness of fit for observations closest to harvest was 35 and 29% nRMSE for fertilisation application rate F1 and F2, respectively, and ME was 0.24 for both fertilisation rates (Table S3 in SM). Since ME was above zero, the model corresponded to the observed data more efficiently than the mean observed value (explained in section 2.3.1). The model fit to all observed biomass data for all fields was $r^2 = 0.61$, nRMSE = 49% and ME 0.53 for F1, and $r^2 = 0.71$, nRMSE = 38% and ME = 0.47 for F2 (Table S4 in SM).

Accurate simulation of biomass is important for estimating soil C inputs, which is a crucial driver for simulating soil C change.

Discussion

Climate impact

Assessment of the climate impact categories revealed considerable variation between the study sites. The mean climate impact for all sites was 178 ± 77 kg CO_2-eq Mg DM^{-1} or 1170 ± 460 kg CO_2-eq ha^{-1} y^{-1} and 136 ± 59 kg CO_2-eq Mg DM^{-1} or 1200 ± 460 kg CO_2-eq ha^{-1} y^{-1} for the F1 and F2 fertilisation rate, respectively. The higher fertilisation rate resulted in higher yields, which reduced the climate impact per Mg DM compared with the F1 rate. However, the difference in climate impact between F1 and F2 was small when analysed per hectare. Overall, the site-specific properties were more important than fertilisation rate when assessing the climate impact of grass cultivation (Figure 5). The main emissions causing the climate impact were in the form of soil N_2O emissions and emissions from fertiliser manufacturing, while the increased soil C content reduced the climate impact of the system. Negative CH_4 emissions also contributed to reducing the climate impact, but at a very small scale compared with C sequestration. Soil can act as both a source and sink of CH_4, depending on the soil environment. However, less managed soils such as native prairie and forest soils are normally net consumers of CH_4 (Johnson et al. 2007).

Assessment of the climate impact over time showed lower impact during the first part of the simulation period, when C sequestration was higher and compensated for the impact of other emissions (Table 3). As yearly C sequestration decreased, the climate impact increased, which resulted in increased global mean temperature after both 30 and 100 years (Figure 6). The risk of soil C sequestration schemes transitioning from global warming mitigating to global warming

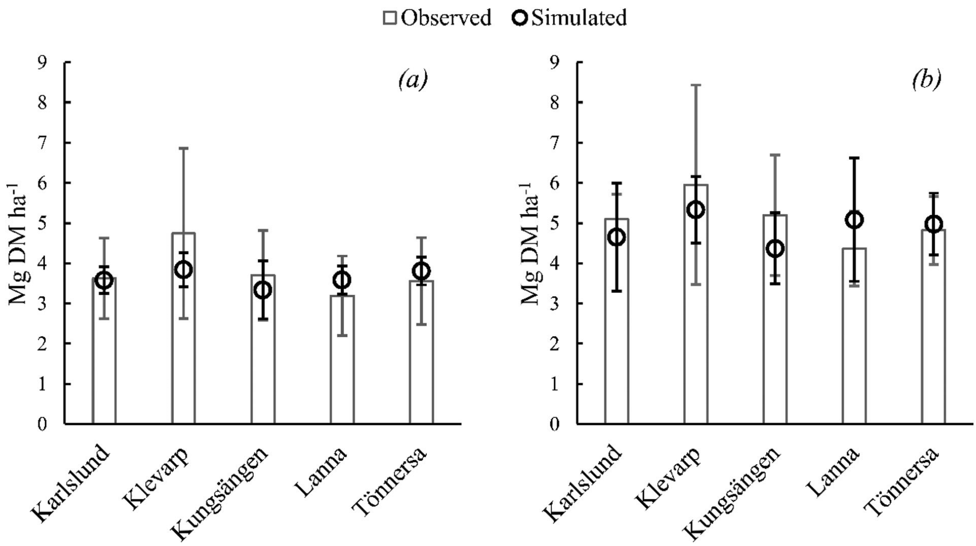

Figure 8. Aboveground mean biomass production at harvest, with fertilisation rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-1}). Bars represent mean of the observations and the rings simulated means. The error bars represent the standard deviation of the observations (grey) and simulations (black). DM = dry matter.

forcing when the soil approaches SOC saturation has been discussed earlier (Lugato et al. 2018). This risk is especially high in agricultural systems that are dependent on mineral fertilisers to maintain SOC content, because of the large climate impact associated with fertiliser manufacturing and enhanced N_2O emissions from soil. Moreover, in terrestrial systems the C and N cycles are closely coupled, which means that a change in C stock will ultimately alter the conditions for N soil processes, such as nitrification and denitrification. Li et al. (2005) investigated the relationship between SOC content and N_2O emissions in both modelling studies and field trials. They concluded that strategies to increase SOC content, such as reduced tillage, enhanced crop residue incorporation and farmyard manure application, increase the N_2O emissions, offsetting the mitigating effect by 75-310% (Li et al. 2005). However, this pattern is not undisputed. For instance, studies in Canada have shown that reduced tillage in dry semi-arid and sub-humid soils can decrease N_2O emissions due to lower nitrification rates in poorly aerated soils, while reduced ploughing in more humid regions can result in increased N_2O emissions (Helgason et al. 2005; Rochette et al. 2008).

Soil C sequestration

The introduction of grass cultivation resulted in increased soil C stock at all sites over the 30-year simulation period. The F2 fertilisation rate induced more C sequestration than F1 (Figure 3). This was because of the increased mean gross C input, which was 2.7 and 3.4 Mg ha^{-1} y^{-1} for F1 and F2, respectively. The greatest increase in soil C stocks was predicted for the silty clay loam at Lanna (0.14 and 0.22 Mg ha^{-1} y^{-1} for F1 and F2, respectively). Goglio et al. (2014) used DNDC to assess soil GHG emissions in an LCA study and concluded that it can accurately simulate C inputs. Furthermore, in a study using dry combustion analysis to determine C sequestration in long-term grass and cereal rotations at two sites, including Lanna, Börjesson et al. (2018) concluded that a mean C input of roughly 2.5 and 3.5 Mg C ha^{-1} y^{-1} increased the soil C stock by 0.11 and 0.17 Mg C ha^{-1} y^{-1}, respectively. The grass rotation in that study included three years of grass-clover mixture and one year of cereals (Börjesson et al. 2018), and thus less C sequestration could be expected since the perennial period was shorter and an annual crop was present in the rotation.

Clay and SOC content are two important soil properties that influence the C sequestration potential. Soils with a high SOC content are usually closer to their C saturation concentration, which means lower capacity to sequester C, while a high clay content affects the decomposition rate by making organic material physically unavailable to the soil decomposers (Li et al. 1992). The effects of the interaction between SOC and clay content are not always trivial. For instance, in the present study C sequestration was estimated to be greatest in the soil with the second highest clay content (33%) and moderate SOC content (2%), while the soil with the highest clay and SOC content (57% and 6%, respectively) had the lowest soil C sequestration under F1 fertilisation (Figure 3). However, the F2 fertilisation rate induced increased soil C stocks in the same soil, by 0.12 Mg C ha^{-1} y^{-1}, due to the increased C input and the high SOC binding capacity associated with the high clay content. The soil with the lowest clay content showed low C sequestration ability, even though the initial SOC content was low.

In this study, we did not include CO_2 assimilated in the biomass yield, which corresponded to 9.8 and 13.2 Mg CO_2-eq ha^{-1} y^{-1} for fertilisation rate F1 and F2, respectively. This means that, although production of grass increased the global mean temperature, there is potential for creating climate-mitigating systems depending on how the harvested biomass is utilised.

Soil N_2O emissions

The yearly cumulative N_2O emissions showed large variation between sites and fertilisation intensities (Figure 4). In general, the N_2O emissions were higher from fine-textured soils than from coarser-textured soils. Soil water content and water-filled pore space have been shown to be appropriate parameters for describing soil redox potential, and thus the conditions for soil N_2O formation (Li et al. 2000). Soils with high water content are often characterised by low redox potential, which favours the formation of N_2O through denitrification. For the DNDC simulations in the present study, data on water retention parameters such as porosity, field capacity and wilting point for each soil were input directly into the model. In contrast, the Rochette *et al.* approach uses soil sand content to describe soil water-filled pore space (Rochette et al. 2018). This explains why the Rochette *et al.* approach gave the highest N_2O emissions for the soil with the lowest soil sand content, while the DNDC model gave the highest emissions for the soil with the greatest water holding capacity (Table 1). Both site-specific methods gave the lowest emissions for the sandy loam soils. When measurements of N_2O emissions are not available, estimation using the IPCC Tier 1 approach is common in LCA studies. For all sites in this study, the Tier 1 approach predicted similar mean N_2O emissions to the other methods tested, which indicates that IPCC Tier 1 could be an adequate tool for estimating mean emissions in site-independent studies. However, since it does not consider how the emission rate is affected by spatial variations, e.g. soil properties and climate, it may not produce reliable results for site-specific LCAs.

Eutrophication

Mean potential eutrophication at the five sites included in the present study was 11.1 ± 6.1 kg N-eq ha^{-1}, while mean marine eutrophication was 4.2 ± 5.4 kg N-eq ha^{-1}. Use of the site-independent CML method to assess the eutrophication potential of the grass cultivation system at different locations in Sweden revealed the most substantial impacts for sandy loam soils, due to the relatively high N leaching rate (Table 4 and Figure 7). One of the advantages of the CML approach is that it includes all types of recipients. The main disadvantage is that it does not consider how eutrophying emissions affect different types of environments. Eutrophication impact is highly spatially dependent and therefore site-specific methods are preferable, especially in regions with complex environments such as the Baltic Sea. The site-specific method used in this study to assess marine eutrophication accounts for site and catchment properties, as well as the limiting nutrient in the recipient. The Henryson *et al*. approach estimated by far the highest marine eutrophication impact for the Tönnersa site, because of high N leaching and the proximity of the site to an N-limited recipient. The other sandy loam soil (Klevarp), also with high N leaching rates, was estimated to have much lower marine eutrophication impact, due to high N retention in freshwater along the transport pathway to the marine recipient. Furthermore, the Henryson *et al*. approach does not include the eutrophication effects on freshwaters, primarily caused by P addition, which is covered with the CML method. These two different methods to assess the eutrophication effect of grass cultivation should not be directly compared, since they are used to assess different types of eutrophication. They should instead be viewed as complements to each other and used to provide a more complete picture of the eutrophication situation of study systems.

Concluding discussion

Climate impact assessment showed substantial variation between five study sites at different locations in Sweden. The mean climate impact was 1170 ± 460 and 1200 ± 460 kg CO_2-eq ha^{-1} y^{-1} for a fertilisation rate of 140 and 200 kg N ha^{-1}, respectively. The difference in climate impact between the two fertilisation rates was greater when expressed per Mg DM (178 ± 77 and 136 ± 59 kg CO_2-eq for F1 and F2, respectively). The climate impact was greatest for a heavy clay and SOC-rich soil, while it was lower for sandy loam and silty clay loam soils. In general, soil properties and weather conditions were more important than fertilisation rate for the estimated climate impact of the system.

The climate impact increased over time, with a low impact during the first part of the simulation period for most fields and an increased impact during the latter part due to decreased C sequestration rate. This pattern was not captured with the GWP method, which does not account for the timing of emissions.

There were only small differences in the results when overall mean N_2O emissions were compared between modelling approaches. However, the two site-specific methods, DNDC and Rochette *et al.*, showed large variations between sites, which were not captured with the IPCC Tier 1 approach. The DNDC model predicted the highest emissions for the soil with the highest water-holding capacity, while the Rochette *et al.* approach predicted the highest emissions for the soil with the lowest sand content. This was due to their different inherent approaches to estimating water-filled porosity in soil. Both site-specific methods predicted the lowest emissions from sandy loam soils.

Mean potential eutrophication estimated with the CML method was 11 ± 6.1 kg N-eq ha^{-1}, with the high standard deviation indicating considerable variation between sites. Potential eutrophication was highest for sandy loam soils and lowest for soils with a higher clay content. Marine eutrophication assessed with a site-specific method was greatest for a sandy soil with high N leaching rate at a site in close proximity to the recipient.

Simulation of grass cultivation is known to be complex, primarily because grasses are generally grown in a mixture of species. It is difficult to predict how the proportions of the species vary between years and locations. In the model set-up for this study, the grass mixture was simulated as one crop. Despite this, the DNDC model managed to reproduce observed biomass growth with positive model efficiency values, both for all observations and for observations closest to harvest (Figure 8 and Table S3).

Overall, the great variation found between sites in this study stresses the importance of including temporal and spatial dependency in agricultural LCAs. When important data are lacking, agro-ecosystem models such as DNDC can be a useful tool in completing the life cycle inventory.

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Disclosure statement

No potential conflict of interest was reported by the author(s).

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Regional variation in climate impact of grass-based biogas production: A Swedish case study

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Transitioning from a fossil economy to a bio-economy will inevitably increase the demand for biomass production. One strategy to meet the demand is to re-cultivate set-aside arable land. This study investigated the climate impact and energy potential of grass-based biogas produced using fallow land in Uppsala municipality, Sweden. The assessment was performed on regional level for more than 1000 individual sites, using the agro-ecosystem model DeNitrification DeComposition (DNDC) in combination with time-dynamic life cycle assessment methodology. The results showed that the system could significantly increase biogas production within the region, which would reduce the climate impact by 9950 Mg CO_2-eq per year. Compared with diesel fuel, the grass-based biogas gave a GWP reduction of 85%. However, the site-specific GWP reduction showed large spatial variability, ranging between 102 and 79% compared with diesel fuel, depending on where in the region the grass was cultivated. Two alternative scenarios were investigated, increased mineral N fertilisation and inclusion of N-fixing crops in the feedstock mixture. The highest mitigation per biogas energy produced was found for the N-fixing scenario but, because of lower yields, this scenario had lower total mitigation potential for the region than the increased fertilisation scenario. The increased fertilisation scenario had a lower climate mitigation effect per biogas energy produced, but the highest mitigation potential when the whole region was considered, because of the increased biogas production. The method applied in this study can guide land-use planning of local energy production from arable land, also for other regions.

1. Introduction

To avert the most critical harms of global warming, the world must promptly reduce greenhouse gas (GHG) emissions overall and, in particular, from fossil sources (IPCC, 2014). One strategy to phase out fossil sources is to replace them with bio-based alternatives, thus transitioning from a fossil economy to a bio-economy. This transition will inevitably increase the demand for biomass production (Lewandowski, 2015). This increasing demand can partly be met by re-cultivating set-aside arable land, which has low short-term competition with food production and has less impact than conversion of natural land (Tilman et al., 2009). In Sweden, a major challenge in the transition to a fossil-free economy is the transport sector, where about 77% of the energy use is fossil-based (SEA, 2019). The future demand for biofuels is projected to constitute about half the energy use in the sector, both in the intermediate and long-term perspective (SOU, 2013). Biogas is a competitive biofuel option, generated from anaerobic digestion typically of organic wastes, such as food waste and sewage sludge. The produced biogas can replace fossil energy in power and heat generation as well as in transportation. Furthermore, biogas is a storable energy carrier that can be saved for future energy use (Weiland, 2010), and may therefore fit well into energy systems with large shares of renewable intermittent energy sources. In 2017, Swedish production of biogas was 7.6 PJ, of which about two-thirds were upgraded to vehicle fuel, mainly used as fuel for cars and buses. In the same year, the total amount of fuel delivered amounted to 333 PJ (SEA, 2019). Besides energy, the digestate produced in the biogas process can be used as organic fertiliser, reducing the demand for mineral fertiliser and adding carbon (C) to the soil.

Soil C sequestration has been advocated as a cost-effective

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strategy with high potential to mitigate global warming. Soil C is more abundant in perennial cropping systems, owing to greater root biomass production, less exposure to soil disturbance and longer growing seasons (Bolinder et al., 2010). Hammar et al. (2017) showed that willow grown on fallow land in Sweden could generate energy and simultaneously remove C from the atmosphere through enhanced soil C sequestration. One of the most common perennial crops in Sweden is grass, which occupies about 40% of the total arable land (SCB, 2018b). Grass is grown worldwide mainly for fodder, but alternative uses such as feedstock for bioenergy purposes are becoming more common (Carlsson et al., 2017).

Life cycle assessment (LCA) is a quantitative method for studying the environmental burden of products and services in a life cycle perspective, from cradle to grave. The method was initially developed as a site and time-independent tool for industrial systems but, over time, has become applicable for other types of systems. For LCAs involving agricultural processes, spatial and temporal dynamics could have a significant impact on the total environmental performance. For example, the GHG balance is heavily dependent on properties such as soil type, climate and agricultural practices (Miller et al., 2006). However, LCA studies that include fine-scale spatial differentiation over time and space are quite rare, due to the large data demand (Nitschelm et al., 2016).

Previous studies have shown that agro-ecosystem models can be used in LCAs to generate site-specific data (Goglio et al., 2018b, 2014). In an earlier study (Nilsson et al., Unpublished results), we combined LCA methodology and the agro-ecosystem model DNDC to assess the environmental impact of grass cultivation at five sites in Sweden. In the present study, we extended the system to grass-based biogas production on regional level, using set-aside arable land in Uppsala municipality, located in east-central Sweden. In Uppsala municipality, about 10% of total arable land is reported to be under fallow (SCB, 2018a), of which more than 50% has been unused for more than three consecutive years (SCB, 2017).

The overall aim of this study was to assess the energy potential and climate impact of converting current unused arable land in Uppsala municipality to intensified grass cultivation and using the harvested biomass to produce biogas. The investigation was performed on a regional level, using existing site-differentiated data. The GHG balance was investigated for each study site, including changes in the soil C stock. The climate impact of the fuel produced (MJ^{-1}) was compared with that of diesel fuel, while accounting for the higher energy efficiency in a diesel engine. Moreover, the climate impact variation within the region was analysed, as was the effect of choosing the most suitable sites.

2. Method and materials

2.1. System boundary

The system boundary included grass cultivation, biogas production, digestate use and biogas use. The grass cultivation was assumed to be located on mineral soils under fallow in Uppsala municipality. The assessment was performed over a 100-year time horizon, which corresponded to 20 grass rotations. Any other co-substrates mixed in the digester were outside the system boundary for this study. Since the land was assumed to be initially unused, no indirect land-use changes were accounted for. The direct land-use effects were defined as the impact of transferring the land from the reference land use (fallow) to the altered land use (grass cultivation) throughout the investigated time horizon. Expansion of infrastructure, such as construction and manufacturing of trucks and machinery and other capital goods were not included in the assessment since their climate impact has been demonstrated to be of minor importance compared to other activities in the system (Hijazi et al., 2016; Tidåker et al., 2016a). All major fluxes of the three main GHGs (CO_2, CH_4 and N_2O) were included in the climate impact assessment. The system was analysed in terms of three different units: (i) hectares (ha) of land, (ii) all investigated fields in Uppsala municipality and (iii) biogas energy produced (MJ). The ha-based unit was used in the inventory analysis to show the effect of land-use change, the field-based unit was used to show the climate impact of increased biogas production in the municipality using fallow land for biogas production, and the biogas-based unit was included to provide figures comparable with results from other bioenergy studies.

2.2. Study region

The study region, Uppsala municipality, is located in east-central Sweden. Information about current land use was obtained from the Swedish Board of Agriculture. The reported fallow land in the region in 2014 was 1977 sites, with a total area of 3587 ha. Organic soils (soil organic matter (SOM) > 20%) and sites smaller in area than 0.5 ha were omitted from the study, which reduced the number of sites to 1240, with a total area of 3006 ha. Fine-textured soils such as silty clay loam, clay loam, silty clay and clay together constituted about 90% of the total area assessed, while more coarse-textured soils were less frequent. The soil C content showed considerable variation, ranging between 0.7 and 11.5%, with a median value of 2.2%. The distribution of soil texture and C content is shown in Fig. S1 in Supplementary Material. The soil pH value ranged from 5.1 to 8.3, with a median value of 6.5. The weather data used consisted of a 10-year sequence, collected between 2007 and 2016, which was repeated in the model within the temporal boundary of the system studied. Mean annual precipitation for this period was 596 ± 77 mm, and mean annual temperature was 6.5 ± 0.9 °C. We assumed the same location for the biogas plant as for the current largest existing plant in the region (Fig. 1).

2.3. System description

The studied system was divided into six subsystems: grass cultivation (*GrassC*^{A}), biomass conversion (*BioC*^{A}), digestate (*Dig*^{A}), fallow (*Fall*^{R}), fossil fuel (*Foss*^{R}) and mineral fertiliser (*Min*^{R}) (Fig. 2).

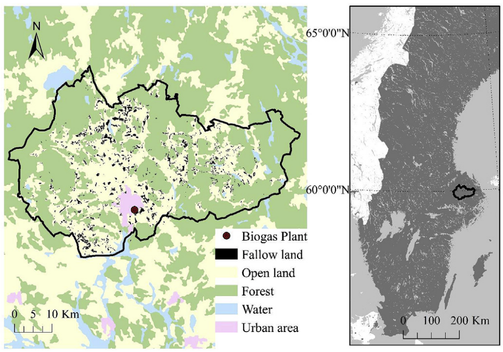

Fig. 1. (Left) Map of the study region, Uppsala municipality (inside the black line), showing the distribution of fallow land (black dots) and the location of the biogas plant (red and blackpurple dot). (Right) Location of the region in east-central Sweden. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

The first three subsystems comprised the altered system (A) and the latter three the reference system (R). The subsystems were also clustered into three compartments, land use (ΔLU), fuel production (ΔFP) and soil fertilisation (ΔSF), to assess the net impact of the different steps in the life cycle. The emissions (*E*) from ΔLU were assessed as the difference between *GrassC^A^* and *Fall^R^*, those from ΔFP as the difference between *BioC^A^* and *Foss^R^* and those from ΔSF as the difference between *Dig^A^* and *Min^R^*. The basis of the comparison in the ΔLU compartment was area, i.e. the calculated emissions were based on the same area of grass cultivation and fallow. For the ΔFP compartment, engine energy was the basis for comparison, while for the ΔSF compartment it was nitrogen (N) uptake. The total GHG emissions (E_{Tot}) were calculated as the difference between the altered system and the reference system as:

$$E_{Tot} = \overbrace{\left(E_{GrassC^A} - E_{Fall^R}\right)}^{E_{\Delta LU}} + \overbrace{\left(E_{BioC^A} - E_{Foss^R}\right)}^{E_{\Delta FP}} + \overbrace{\left(E_{Dig^A} - E_{Min^R}\right)}^{E_{\Delta SF}} \quad (1)$$

2.3.1. Land use

The net climate impact from ΔLU was assessed by subtracting the impact of *GrassC^A^* from the impact of the *Fall^R^* subsystem (Fig. 2).

In *GrassC^A,^* the grass, a mixture of timothy (*Phleum pratense* L.) and meadow fescue (*Festuca pratensis* Huds.), was grown in five-year consecutive rotation periods. The rotation started with sowing and rolling in the first year and ended with ploughing. During the rotation period, the grass was cut, chopped and fertilised with mineral fertiliser twice a year. In total, 140 kg N fertiliser were applied per ha and year. At each cut, 85% of the aboveground biomass was assumed to be harvested.

Diesel consumption for sowing, rolling and spreading fertiliser was set to 2.3, 2.3 and 4.7 dm^3 ha^{-1}, respectively, whereas diesel consumption for cutting, chopping and ploughing was based on linear regression models with biomass yield and clay content as the independent variable (Eq. S1). The GHG emissions from mineral fertiliser manufacturing were 3.6 kg CO_2-eq kg N^{-1}, where the climate impact was set to 86% from CO_2 emissions, with the remaining 14% from N_2O (Brentrup et al., 2016).

The fallow land was assumed to be covered with vegetation, so-called green fallow. The only field operation conducted on the fallow land was cutting, which was performed once a year during late autumn. The cut biomass was left in the field.

2.3.2. Fuel production

The net climate impact from ΔFP was calculated as the difference between *BioC^A^* and *Foss^R^* (Fig. 2). After each cut, the harvested feedstock was transported to the biogas plant with freight trucks. The energy consumption for using a truck with trailer, load capacity 34–40 Mg, was taken from https://www.transportmeasures.org. The energy use per transport Mg x km was 1 MJ, including empty positioning of the truck.

At the biogas plant, the harvested biomass was loaded into bunker silos. The diesel consumption for biomass compaction in the silo was calculated based on the weight of the compressed biomass (Eq. S2). Biogas energy produced was derived based on the amount of biomass added to the biogas reactor and the specific CH_4 production, 280 Nm^3 Mg VS^{-1}, where the volatile solids (VS) content was set to 92% of dry matter (DM). The ensiled biomass was continuously fed to the biogas reactor, where mesophilic anaerobic digestion converted the biomass to biogas that was upgraded to bio-methane. A part of the biogas produced was used to heat the reactor. The biogas conversion processes pumping, stirring, upgrading and gas compression were all considered to be electrically driven. Emissions and primary energy use for the electricity were assessed using data for the Nordic electricity mix, which was assumed to be close to the expansion margin based on the stated goal of a continuous high share of renewables in the Swedish electricity mix (Government Offices of Sweden, 2016). After the digestion, the digestate produced was assumed to be stored, before being transported to farms and spread in winter wheat cultivation.

The CH_4 losses from biomass conversion were assessed using data from the existing plant in Uppsala for 2015, when measured losses during anaerobic digestion and upgrading with water scrubbers were 0.01% and 0.3% of methane production, respectively (Uppsala Vatten, 2017). The losses from digestate storage were calculated using the equation for large and medium-sized biogas plants given by Styles et al. (2016) (Eq. S2).

The biogas produced was assumed to replace diesel fuel, *Foss^R^*. In the calculations, the higher efficiency in the diesel engine was considered by using an energy efficiency of 9.8 MJ km^{-1} for the diesel compared with 11.4 MJ km^{-1} for the biogas (Börjesson et al., 2016). Hence, one MJ biogas replaced 0.86 MJ of diesel.

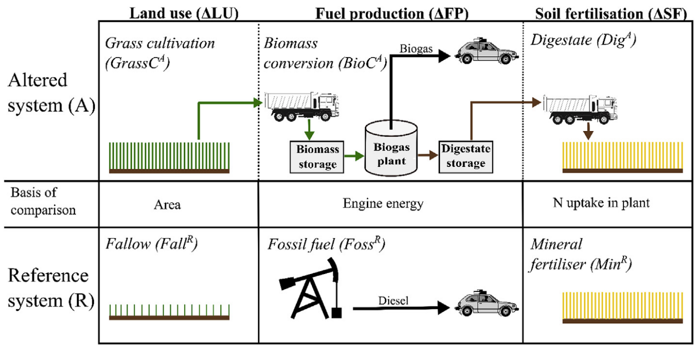

Fig. 2. Schematic illustration of the grass-based biogas system studied, divided into six subsystems: Grass cultivation ($GrassC^A$), Biomass conversion ($BioC^A$), Digestate use (Dig^A), Fallow ($Fall^R$), Fossil fuel ($Foss^R$) and Mineral fertiliser (Min^R). The net effect of the system was calculated as the difference between altered system and reference system. The subsystems were also divided into three compartments: Land use ($GrassC^A$ - $Fall^R$), Fuel production ($BioC^A$ - $Foss^R$) and Soil fertiliser (Dig^A - MinR). The basis of comparison is shown in the row between the altered system and the reference system.

2.3.3. Soil fertilisation

The net effect of the ΔSF compartment was calculated by subtracting the GHG emissions occurring in winter wheat cultivation with mineral fertiliser (*Min*^{R}) from the emissions from winter wheat cultivation with digestate fertiliser (*Dig*^{A}) (Fig. 2).

The digestate was transported once a year to the winter wheat sites. The distance to the winter wheat cultivation was set to 20 km based on the mean distance to the fallow land from the biogas plant. At pick up, the DM content was 9.5% for the digestate. All transport was performed with the same type of truck as in *BioC*^{A}.

For the cultivation with mineral fertiliser, the amount of N applied was 135 kg ha^{-1}. The same spreading technique was used as for the grass cultivation subsystem. The amount of digestate produced in the system was calculated by following the mass balance from biomass input to the reactor to field application (Fig. 3).

The C and N content of the digestate was obtained by calculating the C losses in the form of CO_2 and CH_4 conversion during anaerobic digestion and CH_4 emissions during the digestate storage phase. The biogas before upgrading was assumed to contain 55% CH_4. The N content in the digestate at application was assessed by calculating the losses of N, in the form of N_2O and NH_3, during digestate storage (Fig. 3). The equation used to calculate the conversions is presented in Supplementary Material (Eq. S3).

In order to compare the digestate to the mineral fertiliser, the mineral fertiliser equivalent (MFS) was calculated to represent the difference in fertilisation effect, i.e. how much digestate was needed to replace the mineral N, given the specific composition of the digestate. The MFS was obtained by iteratively executing the agro-ecosystem model (section 2.4.2) with different amounts of applied digestate until the average yields corresponded. The MFS for the digestate produced was found to be 80%, leading to a total amount of digestate spread per hectare of 37.1 Mg (wet weight), containing 1183 kg C and 169 kg N (tot-N). The digestate properties are presented in Table S1. The diesel consumption for spreading the digestate was 0.31 dm^3 MJ^{-1}.

2.4. Life cycle inventory analysis

2.4.1. GIS model

The ArcGIS product (ArcMap version 10.3, ESRI, Redlands, CA, USA) was used to link soil data to the specific study sites in the region. All land reported as being under fallow was linked to specific soil properties, in terms of initial soil organic matter, clay, silt and sand content and pH. This was done by interpolating data from 258 measurement points spread out over the study region. ArcGIS was also used to calculate road route distances from the grass cultivation sites to the biogas plant.

2.4.2. Agro-ecosystem modelling

The process-based agro-ecosystem model DNDC (DeNitrification DeComposition) was first developed in 1992 to model C and N

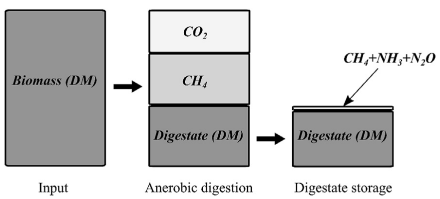

Fig. 3. Conceptual model of the mass balance calculation for digestate (illustration not to scale).

fluxes in agricultural soils (Li et al., 1992). Since then, the model has been updated and branched into several versions, which have been used in studies all over the world (Gilhespy et al., 2014). In the present study we used the Canadian version, DNDC-CAN, which has been validated in similar cool-weather conditions as those prevailing in Sweden. Following an assessment of different methods for estimating soil-borne N_2O and CO_2 emissions, Goglio et al. (2018a) concluded that DNDC was the only model among those tested that gave similar results to measurements for N_2O emissions estimates. Here, the model was used to generate annual, site-specific, inventory data comprising biomass yields, soil C balances and soil N_2O and CH_4 emissions. The input variables field capacity, wilting point porosity and bulk density were estimated using a pedotransfer model developed by Saxton and Rawls (2006).

The crop and management model set-up was the same as in Nilsson et al. (Unpublished results), in which the same grass mixture was modelled at five locations in Sweden. The fallow land was simulated with the same set-up as for the grass crop, but without added fertiliser. In order to capture the initial effect of the grass-based biogas system, the simulation was formulated to include a spin-up period with the reference system land use, which was executed before collection of the inventory data started. We used a spin-up period of 10 years, which is typically used for the DNDC model (Grant et al., 2016).

The effect of using the digestate as fertiliser was analysed by executing the DNDC model for winter wheat cultivation, both with digestate application and mineral fertiliser. In contrast to the *GrassC*^{A} subsystem, which was modelled for all 1240 fields, *Dig*^{A} was modelled for one field which represented the average conditions in the region. Both fertiliser options were assessed with the same management procedure, in terms of timing for ploughing, harvesting and spreading fertiliser. The winter wheat area for which the N demand could be met by the digestate produced from 1 ha of grass cultivation, here denoted F_{dig}, was calculated as:

$$F_{dig} = \left(N_{dig} - N_{NH3\ loss\ app}\right) \Big/ N_{demand} \tag{2}$$

F_{dig} was multiplied by the GHG emissions per hectare for the simulated winter wheat cultivation to obtain the GHG balance from the *Dig* subsystem, where N_{dig} (kg N ha^{-1}) is the N in the digestate, $N_{NH3\ loss\ app}$ (kg N ha^{-1}) is the N–NH_3 losses during digestate application, and N_{demand} (kg N ha^{-1}) is the N demand of winter wheat, i.e. the amount of mineral N applied divided by MFS (explained in section 2.3.3). Model input parameters for all the different land uses are listed in Table S2.

2.4.3. Energy conversion

The energy output from the altered system was calculated at regional level. The major primary energy input, in terms of fossil fuel and electricity, was included. The biogas produced was assumed to be partly used to heat the biogas plant, so heat was not considered an energy input. The energy performance of the altered system was finally assessed by calculating the energy ratio (ER) (Djomo et al., 2011), calculated as the ratio of energy produced to primary energy input:

$$Energy\ ratio = Energy\ output / Primary\ energy\ input \tag{3}$$

2.5. Climate impact assessment

The climate impact was assessed both with GWP methodology and with Absolute Global Temperature Potential (AGTP), defined by Myhre et al. (2013). The latter approach is used to assess the temperature response, in Kelvin (K), to changes in radiative forcing caused by GHG fluxes. All GHGs have different impacts on radiative forcing, depending on atmospheric lifetime and radiative efficiency, i.e. the impact on the balance of incoming solar and outgoing terrestrial radiation. The annual net fluxes of all major GHGs (CO_2, CH_4 and N_2O) from the system were annually aggregated and converted to temperature response over the analytical time horizon, 100 years. This rather extensive time frame was adopted to enable investigation of the time dynamic variation of the climate impact of the system. The temperature response for each year was then accumulated for each of the simulated years as:

$$\Delta T_i(H) = \sum_{t=0}^{H} X_i(t) AGTP_i(H-t) \quad (4)$$

where $\Delta T_i(H)$ is the cumulative temperature response to the flux of GHG i during analytical time horizon H, $X_i(t)$ is the total flux of GHG i in year t, and $AGTP_i(H - t)$ is the temperature response of GHG i flux between the time t and the analytical time horizon H per unit GHG. This approach can be used to assess the dynamic climate impact and has previously been used in LCA studies to evaluate the climate impact of bioenergy systems (e.g. Hammar et al., 2017).

A more common approach to assess the climate impact is determination of Global Warming Potential (GWP), where the radiative forcing caused by a pulse emission of a GHG is calculated and compared with the same amount of CO_2 over a specific time horizon, normally 100 years. In contrast to the dynamic AGTP approach, GWP does not include the timing of the GHG flux, which means that emissions that occur during different points in the life cycle are added together, although the endpoint of the impact differs (Kendall, 2012). The characterisation factors used here in GWP calculations were 34 and 298 for CH_4 and N_2O, respectively, with the inclusion of climate-carbon feedbacks (Myhre et al., 2013). The net GWP for the biogas produced, without fossil fuel substitution, was compared to diesel by calculating the GWP reduction from replacing the fossil alternative with the biogas:

$$GWP\ reduction = (GWP_F - GWP_B) / GWP_F \quad (5)$$

where GWP_B is the GWP caused by net emissions from the system under study, without fossil fuel substitution (i.e. $E_{Tot} - E_{Foss^R}$), and GWP_F is the GWP caused by emissions from an equivalent amount of fossil fuel (E_{Foss^R}).

PE input (TJ y^{-1})	
Grass cultivation (*GrassC^A*)	
Ploughing	0.5
Sowing	0.1
Rolling	0.1
Cutting	0.8
Chopping	2.8
Fertiliser manufacturing	14.8
Fertiliser spreading	1.1
Biomass conversion (*BioC^A*)	
Grass transport	1.2
Compaction	1.3
Pumping stirring	5.5
Upgrading	9.1
Compression	7.9
Digestate use (*Dig^A*)	
Transport digestate	1.7
Application	0.9
Total PE input	*47.8*
Energy output (TJ y^{-1})	
Biogas (fuel)	167.4
Total energy output	*167.4*
Energy ratio (GJ GJ^{-1})	3.5

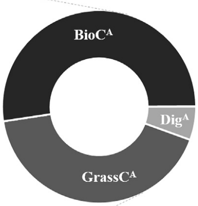

Fig. 4. Annual primary energy (PE) input and energy output of the altered system for the study region, divided between the subsystems grass cultivation (GrassC^A), biomass conversion (BioC^A) and digestate use in winter wheat cultivation (Dig^A).

2.6. Alternative scenarios

Two alternative scenarios were compared with the base scenario, the grass-based biogas system described in section 2.3. These were: (i) increased mineral fertilisation rate in the *GrassC^A* subsystem, from 140 to 200 kg N ha^{-1} and (ii) exclusion of all mineral fertiliser in the *GrassC^A* subsystem based on the assumption that the feedstock crop can satisfy its N demand through biological N fixation from the atmosphere, e.g. a grass-clover mixture. Both alternative scenarios were simulated in DNDC, with otherwise the same model set-up. For the N fixation scenario, the fixation rate was adjusted so that the average yield was about 15% lower than for the base scenario (Tidåker et al., 2016b).

3. Results

3.1. Inventory results

3.1.1. Energy balance

The annual primary energy input and energy output from the altered system are shown in Fig. 4. On a yearly average using all 1240 land sites, the vehicle fuel produced amounted to 167 TJ biogas y^{-1}, with a primary energy input of 47.8 TJ. This resulted in an energy ratio of 3.5, which means that for every energy unit input in terms of fossil fuel and electricity, the system produced 3.5 units of biogas fuel. The highest primary energy input was in *BioC^A*, where upgrading and compression were the processes with the highest energy use. For *GrassC^A*, most energy input was required for manufacturing the mineral fertiliser, which represented 31% of the total energy input. The energy gained from replacing mineral fertiliser with digestate was not included in the energy balance.

3.1.2. Soil carbon balance

The modelled soil C balance of *GrassC^A* showed large differences between the different sites (Fig. 5). In the field with the highest ability to sequester C, the stock was increased by 16 Mg C ha^{-1} during the study period, which corresponded to a sequestration rate of 160 kg C ha^{-1} y^{-1} averaged over the simulated 100 years. The C sequestration rate was higher during the first part of the period than in the latter part. This pattern was more evident in the soil with the median change, 6 Mg C ha^{-1}, where the C stock reached equilibrium in the first half of the simulated period. The site with the lowest ability to sequester C lost 13 Mg C ha^{-1}. Large variation between the sites was also seen for the *Fall^R* subsystem (Fig. 5). Compared with the gross effect of *GrassC^A* and *Fall^R*, the net effect of ΔLU showed lower spatial variability, ranging between 10 and 4 Mg C ha^{-1} with a median increase of 6 Mg C ha^{-1}. The net effect indicated an increased soil C stock at all sites, which means that 100 years of grass cultivation resulted in a larger soil C stock in the region than continued fallow land.

The soil C balance was further investigated by simulating the effect of digestate use on soils with median soil properties in the region. The use of digestate in winter wheat cultivation increased the soil C stock while the mineral fertiliser showed depletion, which entailed a large C increasing net effect of the ΔSF of 23 Mg C ha^{-1} (Fig. 6). On average, the mean digestate produced per ha grass cultivation covered the N demand of 0.66 ha of winter wheat cultivation.

The correlations between input data soil properties and the soil C sequestration potential in *GrassC^A* were analysed using Pearson correlation coefficient (r) (Table S3). The strongest correlation was found for initial C content, which had a negative correlation with

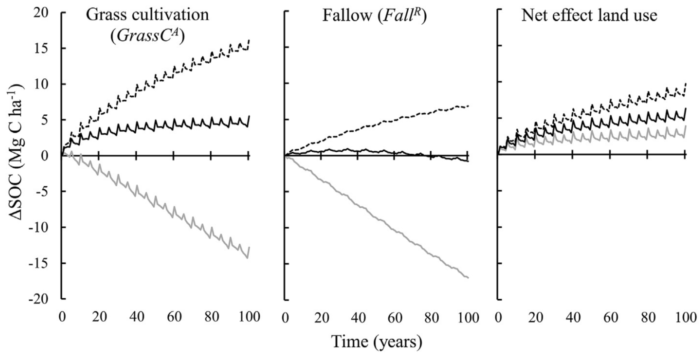

Fig. 5. Cumulative change in soil organic carbon (SOC) over 100 years for all sites investigated (N = 1240), simulated with the DNDC model, for (left) grass cultivation only, (centre) fallow land only and (right) the net effect of changing the land use from fallow to intensified grass cultivation. The dashed black line represents the 95th percentile (max), the grey line the 5th percentile (min) and the black line the median.

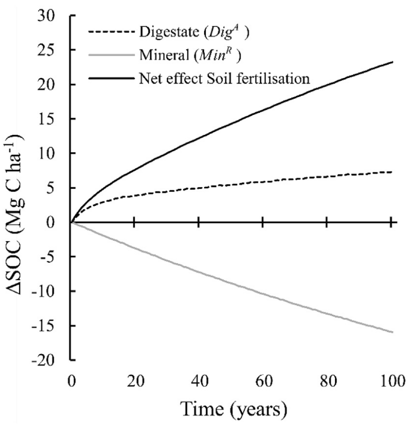

Fig. 6. Cumulative change in soil organic carbon (SOC) over 100 years, simulated with DNDC model, for winter wheat cultivation with biogas digestate as fertiliser (dashed), mineral fertiliser (grey) and the net effect, i.e. the difference between digestate and mineral fertiliser (black). The DNDC model was executed with the input parameters setup that represented the average conditions in the region.

cumulative change in C content. The second highest correlation was for clay content, which had a positive correlation with cumulative change in C content.

3.1.3. Soil nitrous oxide emissions

The modelled soil N_2O emissions also displayed large variations between different sites and years. In general, N_2O emissions from *GrassC^A* were higher than from *Fall^R* (Fig. 7). This entailed a mean increased soil N_2O net effect, which ranged between 2.0 and 0.2 kg N_2O ha^{-1} y^{-1}, with 1.3 kg N_2O ha^{-1} y^{-1} from the median soil.

For the ΔSF compartment, the net N_2O emissions were low or negative during the earlier part of the study period and increased over time. The mean net N_2O emissions from the soil fertiliser compartment were 0.50 kg N_2O ha^{-1} y^{-1}, i.e. the digestate application in the winter wheat cultivation increased the emissions of N_2O compared with mineral fertiliser. The soil N_2O emissions in *GrassC^A* had the highest correlation with soil pH, which showed a negative relationship. The second most influential parameter was the initial C content, which showed a positive relationship (Table S3).

3.2. Climate impact assessment

The climate impact assessment revealed a net decreased temperature response over the study period (Fig. 8). Although the altered system entailed an increased temperature over the time horizon studied, the impact was far lower than that from the reference system. This was largely attributable to the substitution of diesel fuel. The increased soil C stock in the ΔLU compartment was not large enough to compensate for other emissions in the subsystem, primarily because of the emissions from fertiliser manufacturing and the elevated soil N_2O emissions from fertiliser usage. The net effect from the ΔSF compartment was a negative temperature response due to the net increase in the regional soil C stock together with the substitution of mineral N fertiliser.

For the altered system, the impact was dominated by emissions from the *GrassC^A* and the *BioC^A* subsystems. In the short-term, the emissions from *BioC^A* determined the magnitude of climate impact. However, over time, the impact of the *GrassC^A* became increasingly significant. This was because the principal GHG emitted from biomass conversion was CH_4, through losses during biogas processing and digestate storage, where about 60% of the CH_4 emissions were from losses during digestate storage. Methane is a relatively short-lived climate forcer, which explains the declining climate impact rate over time (Fig. 8).

For all sites in the region, the net GWP of the biogas produced without fossil fuel substitution (*Foss^R*) was 10 g CO_2-eq MJ^{-1}, which corresponded to a GWP reduction of 85% compared with diesel fuel. When only the best-performing sites from a climate change perspective were selected, the GWP reduction compared with the fossil alternative increased. The total GWP reduction in relation to the fraction of study region land used in the biogas system is shown in Fig. 9. For instance, if only 10% of the best-performing sites were included, the GWP reduction increased to 95%.

The spatial difference in the GWP reduction was further investigated (Fig. 10). The impact varied between −1 and 14 g CO_2-eq MJ^{-1} in the study region, which corresponds to a GWP reduction of 102 to 79% compared with diesel. The variation could at large be explained by differences in net soil N_2O emissions, r = 0.97, which in turn were most affected by soil pH (Table S3). In contrast, net

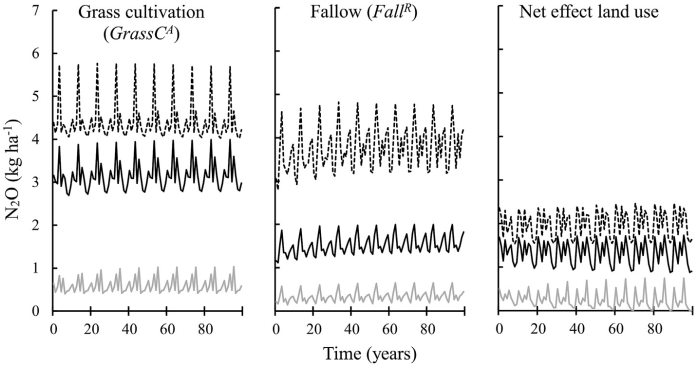

Fig. 7. *Annual soil nitrous oxide (N_2O) emissions for (left) the grass system and (centre) fallow land, and (right) net emissions for feedstock cultivation during 100 years for all sites investigated (N=1240). The dashed black line represents the 95th percentile soil (max), grey line the 5th percentile soil (min) and the black line the median soil.*

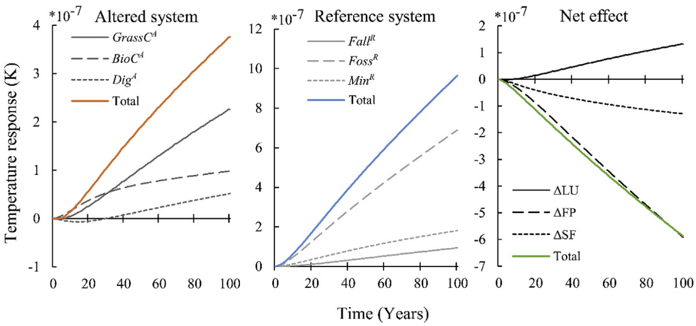

Fig. 8. Temperature response, in degrees Kelvin (K) and using all fields studied (N = 1240, 3006 ha) in the region, for (left) the altered system and (centre) the reference system, and (right) the total net effect.

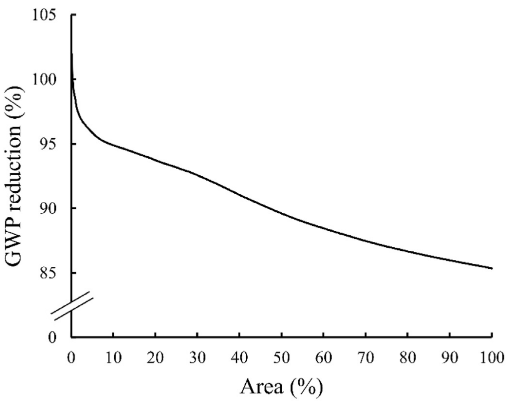

Fig. 9. Global warming potential (GWP) reduction compared with diesel from using the grass-based biogas system, without fossil fuel substitution ($Foss^R$), in relation to fraction of total area used in the region.

changes soil C stock had a low impact on the spatial variation in climate impact (r = −0.28).

3.3. Climate impact of alternative scenarios

The climate impact of the grass-based biogas system and that of the two alternative scenarios (increased fertiliser intensity in $GrassC^A$ and $GrassC^A$ with biological N-fixation) are shown in Fig. 11. The temperature response of the different scenarios was assessed both per biogas energy produced (MJ) and for all fields investigated in Uppsala municipality. The biogas produced in the scenario with increased fertilisation rate showed the lowest climate change mitigation per MJ, −3.4 K$*10^{-17}$, which was similar to that in the base scenario, −3.5 K$*10^{-17}$. The scenario with biological N fixation produced the biogas with the highest mitigation per MJ, −4.6 K$*10^{-17}$. However, due to the assumption of lower yields, this scenario had the lowest overall biogas production, which resulted in lower total mitigation potential for the study region compared with the increased fertilisation scenario. In contrast, the increased fertilisation intensity scenario entailed greater biogas production, which led to the highest climate change mitigation potential for the region. Both alternative scenarios showed greater potential for climate change mitigation in the study region than the base scenario.

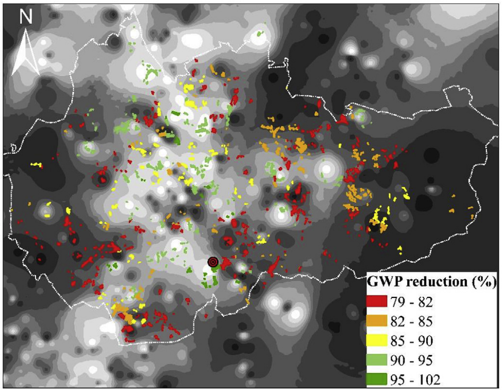

Fig. 10. Spatial variation in global warming potential (GWP) reduction compared with diesel of using the grass-based biogas system, without fossil fuel substitution (Foss[R]). Colours indicate site-specific GWP reduction. Background indicates the soil pH, where a darker shade indicates lower pH. The white dashed line represents the municipality border. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

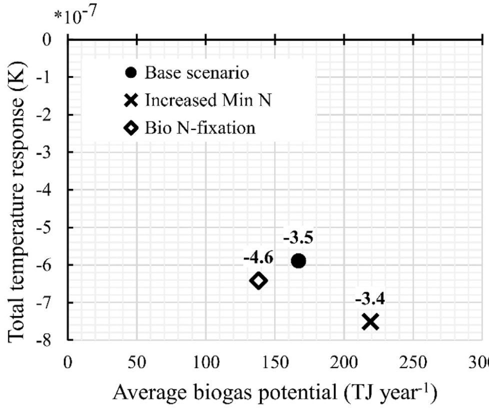

Fig. 11. Total temperature response (degrees K), over 100 years, and average biogas potential (TJ per year) for the base scenario and for two alternative scenarios: increased fertilisation and use of biological N-fixing crops. The numbers next to the icons show the temperature response per unit of biogas produced ($K*10^{-17}MJ^{-1}$).

4. Discussion

In this study, we investigated the energy potential and climate impact of utilising set-aside arable land in Uppsala municipality to produce grass-based biogas. The system studied showed considerable energy potential, with an annual production rate of 167 ± 14 TJ, which would more than double the current biogas production in Uppsala municipality. Besides biogas, the system also produced digestate that could substitute mineral N fertiliser use corresponding to 1980 ha of winter wheat cultivation. The energy ratio for the biogas system was 3.5 (Fig. 4). Energy ratio for large-scale biogas production typically ranges between 2.5 and 5, without including upgrading (Berglund and Börjesson, 2006). Grass-based biogas is usually at the lower end of this range, because of mineral fertilisation and the need for handling the biomass before anaerobic digestion.

The total regional climate impact of the biogas system showed a lower temperature response compared with the reference system (Fig. 8). Based on GWP calculations, biogas from the study system without fossil fuel substitution had a climate impact of 10 g CO_2-eq MJ^{-1} when all 1240 sites with fallow land (3006 ha) were included. This resulted in a GWP reduction of 85% compared with diesel (Figs. 9 and 10). Consequently, using the biogas produced instead of the fossil alternative would considerably decrease the amount of GHG emissions, by about 9950 Mg CO_2-eq y^{-1}.

The biogas system acted as a net atmospheric sink of C, mostly through C sequestration in the ΔLU and the ΔSF compartments. Soil C sequestration is a time-dependent reversible process, where the intrinsic dynamics are a balance between C input and output. For soils that are in equilibrium, i.e. C input equals C output, an increased input will result in an increased soil C stock. The C stock will continue to increase until the soil reaches a new dynamic equilibrium, with the rate of increase normally being faster at the beginning and then levelling off (Kätterer et al., 2012). The temporal aspect is therefore essential when including C balance in a climate impact assessment, as demonstrated by the simulated soil C balance in the present study (Figs. 5 and 6). For instance, the C sequestration rate in grass cultivation with the median soil C change was about five-fold higher in the first 10 years than when averaged over the 100-year study period.

The simulated grass cultivation resulted in a larger gross C stock at most sites investigated (Fig. 5). However, the change in C stock varied between locations. This variation was mainly attributable to initial soil C and clay content, with soils with low initial C stock and high clay content having a greater ability to sequester C. This agrees with findings in previous studies (Bolinder et al., 2010; Poeplau et al., 2015). The simulated C input was quite low on the fallow land, on average 1.7 Mg ha^{-1}. Greater biomass production on this fallow land would reduce the net soil C increase at the sites investigated. Compared with the ΔLU compartment, the net effect of using digestate as fertiliser was a greater net increase in the soil C stock in the ΔSF compartment. This was mainly because of the high C depletion for the winter wheat cultivation with mineral fertiliser. The effects on the soil C balance of using digestate from grass-based biogas production are unfortunately poorly documented. Tatzber et al. (2012) performed long-term field trials of degradation of different organic amendments, e.g. farmyard manure, for which they concluded that the C fraction remaining in the soil after 5, 10 and 37 years was 30%, 20% and 9%, respectively. Using these figures, the soil C sequestration from digestate application would be 10 Mg ha^{-1} over 37 years, which indicates that our estimates may be slightly low.

The simulated soil-N_2O emissions were generally higher for grass cultivation than for fallow land, due to the use of N-fertiliser (Fig. 7). The net N_2O emissions from the ΔLU compartment showed great variation between sites. The strongest correlation to input data was with pH and initial C content (Table S3), indicating that soils with lower pH and high C content generate higher N_2O emissions. Experimental studies have shown that pH affects the ratio between N_2O and N_2 emissions, with increasing N_2O emissions with decreasing pH, which has been attributed to the interference of N_2O denitrification in environments with lower pH (e.g. McMillan et al., 2016; Russenes et al., 2016). Soil N_2O emissions from the ΔLU compartment explained the largest proportion of the net spatial variation in climate impact.

N_2O is a very potent GHG, 298 times stronger than CO_2 over a 100-year perspective. Strategies to increase soil C by intensifying fertilisation may, therefore, be precarious, since soils are not infinite

C sinks. When the soil reaches a new C equilibrium, it will no longer sequester C. However, soil N_2O emissions induced by increased fertilisation rate will continue. Stopping fertilisation at that point would eventually cause lower primary production and hence lower C input, leading to soil C losses. Thus, the effects of increasing the fertilisation rate could go from climate mitigating to climate forcing.

The scenario analysis showed that increasing fertilisation in *GrassC^A* entailed increasing climate change mitigation potential for the study region compared with the base scenario (Fig. 11). This effect was attributed to the increased biogas production, which meant that the system could substitute more diesel fuel. On the other hand, the scenario with biological N fixation displayed the highest climate efficiency, meaning the highest mitigation per biogas energy produced (MJ). The greatest difference in this scenario was that no mineral N fertilisers were added to feedstock cultivation. This reduced the soil N_2O emissions, which is in line with IPCC default values for leguminous crops (IPCC, 2006), where direct N_2O emissions are neglected based on results from Rochette and Janzen (2005). In this study, we added the N-fixing ability to the simulated grass crop in the base scenario, and hence this simulation was not validated against data for N-fixing crops, which needs to be considered when interpreting the results. Because of the lower biogas production, this scenario led to lower mitigation potential than the scenario with increased fertilisation. All the scenarios had a negative temperature response, which meant that the reference system had a larger climate impact than the altered system. However, none of the scenarios achieved negative emissions when only considering the altered system. The lowest temperature response for the altered system was for the N-fixation scenario and the highest was for the increased fertilisation intensity scenario. To increase climate efficiency further, use of fossil fuels in field operations and transport could be excluded and CH_4 losses during digestion and storage could be prevented.

Besides providing a renewable alternative to diesel fuel, the grass-based biogas system investigated here could provide other benefits. For example, cultivating fallow land would increase soil fertility for future biomass cultivation, although of course at the risk of losing the build-up of C stock. The grass biomass produced could also serve as fodder back-up in periods with low fodder production, e.g. due to heatwaves, which are expected to become more frequent with increased global temperature (IPCC, 2014).

Process-based models, such as DNDC, can theoretically be applied to many combinations of geography, climate, cropping systems and management practices. However, existing models are based on the current collective scientific understanding of agro-ecosystem processes and there are still many knowledge gaps that needs to be filled to improve the models. More basic research is therefore essential, e.g. on the processes underlying soil N_2O formation and soil C balance.

5. Conclusions

In this study, biogas production from grass was assessed using LCA methodology in combination with a process-based agro-ecosystem model fed with regional-specific data spatially organised with GIS programming. This combined method could be used to design biomass production schemes in other regions, thereby serving as a strategic tool to assist land use planning of local energy production from arable land. The agro-ecosystem models are, however, limited by scientific understanding of the described processes.

The biogas produced from grass grown on fallow reduced the climate impact significantly, by 79–102%, compared with diesel fuel. Variations in soil N_2O emissions between fields explained most of the spatial variation in climate impact in the study region. By implementing the proposed system, the region's biogas production could on average be doubled, which would reduce the climate impact by 9950 Mg CO_2-eq every year and increase soil fertility in the region through increased soil C stock.

Manufacturing of mineral N fertiliser represented approximately one-third of total primary energy input to the altered system and soil N_2O emissions related to N fertilisation were the greatest source of emissions from the grass cultivation system. Excluding N fertiliser by using feedstock crops relying on symbiotic N fixation, such as clover, increased the energy efficiency and resulted in the highest climate mitigation per energy produce biogas (MJ). However, this scenario reduced biogas production, due to the assumption of lower yields. In contrast, increasing the fertilisation rate in grass cultivation entailed a lower mitigation potential per MJ but higher biogas production, which resulted in the highest climate change mitigation potential in the region.

CRediT authorship contribution statement

Johan Nilsson: Conceptualization, Methodology, Software, Formal analysis, Writing - original draft. **Cecilia Sundberg:** Conceptualization, Writing - review & editing. **Pernilla Tidåker:** Conceptualization, Writing - review & editing, Supervision. **Per-Anders Hansson:** Conceptualization, Writing - review & editing, Supervision, Resources, Funding acquisition.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2020.122778.

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III

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Effect of short-term perennial leys on life cycle environmental performance of cropping systems: An assessment based on data from a long-term field experiment

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ABSTRACT

Modern agriculture's dependence on the intensive use of inputs, such as chemical fertiliser and pesticides, leads to high environmental impacts and, possibly, vulnerability in food security, since most of these inputs are imported from other countries. This calls for more sustainable and resilient agricultural practices. Diversification of crop rotations, e.g. by including perennial leys, enhances provision of ecosystem services, leading to healthier crops and increased yields. Perennial crops also increase soil organic carbon (SOC) stocks, which is interesting from a global warming mitigation perspective. In addition, legume-rich leys can utilise atmospheric nitrogen (N) through symbiotic association with N_2-fixing bacteria. However, few studies have evaluated the effects of short-term perennial leys in rotation on cropping system performance over long periods and under different conditions. In this study, we used data from three sites in a long-term experiment in Sweden (initiated in the 1960 s), in combination with Life Cycle Assessment methodology, to assess the environmental and yield effect of including ley in crop rotations. Two N fertiliser regimes (High, Low) in combination with three six-year crop rotations, consisting of either i) two-year mixed grass-legume ley, ii) two-year pure grass ley or iii) annual crops without ley, were compared. Environmental impacts (climate impact, energy resource depletion, eutrophication potential) of the different combinations were quantified per kg harvested crop (expressed in cereal units, CU) and per hectare. The lowest environmental impact, at all sites, was found for the crop rotation with two-year mixed ley under the Low N regime. On average, this combination resulted in 329 g lower GHG emissions per kg CU than the crop rotation without ley and Low N, primarily due to lower input of chemical N fertiliser, which reduced the impact from fertiliser production and soil N_2O emissions. Comparison of mean SOC change over the study period revealed reduced SOC stocks for all rotations and all sites, especially in the rotation without ley. Therefore, including short-term perennial leys, especially leys containing legume species, in crop rotations can be a useful tool in meeting policy targets on reducing the environmental impacts of agriculture, and in reducing the dependence on purchased agricultural commodities. However, despite the potential benefits of rotational leys, the market demand for the produced ley biomass may be insufficient. Hence, incentives to increase demand are necessary to promote large-scale adoption, for example, for use in bioenergy production and feed.

1. Introduction

Following the *Green Revolution*, chemical fertilisers and biocides have increased food production and helped sustain the growing global population (MacLaren et al., 2022). They have also allowed farmers to specialise in a few crops and abandon the diverse crop rotations that characterised European agriculture since the introduction of perennial grass-legume crops during the 19th century (Mudgal et al., 2010). However, intensive use of inputs in agriculture is known to be directly linked to environmental impacts such as global warming, eutrophication, biodiversity loss and extensive energy use (Campbell et al., 2017; Foley et al., 2011; Stoate et al., 2001; Tang et al., 2021). Reducing the dependence on purchased input commodities could increase cost-efficiency, reduce the environmental impact (Tidåker et al., 2014,

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2016) and enhance regional security of the supply of agricultural goods. The importance of the latter has recently been demonstrated with the invasion of Ukraine, which is causing deep geopolitical disruption in the European area (World Bank, 2022). The resulting high price fluctuations in agricultural input commodities have brought new challenges for farmers, who can no longer rely on business-as-usual. Therefore, new strategies for building resilience to current and future shocks and stresses must be developed, while still avoiding further aggravating upcoming challenges such as global warming.

One potential strategy for reducing the current dependence on agricultural inputs is to promote ecosystem services by re-introducing more diversified crop rotations (MacLaren et al., 2022; Nemecek et al., 2015; Tamburini et al., 2020). Research has shown that diversification of crop rotations can increase nutrient delivery, keep crops healthier, increase yields and reduce yield losses due to weather extremes (Bergkvist and Båth, 2015; Bowles et al., 2020; Gaudin et al., 2015). In particular, the inclusion of perennial crops, such as temporary leys in cropping systems with a high proportion of cereals, has been shown to reduce dedicated pests, with distance in time and space having a large influence on pest occurrence (Kirkegaard et al., 2008). If properly managed, it can also mitigate nitrogen (N) leaching, leading to reduced eutrophication (Larsson et al., 2005).

A key challenge for resilient agricultural systems is to find a sustainable and reliable supply of N. Grass-legume mixtures can provide substantial amounts of N, up to 500 kg N ha^{-1}, with low environmental burden via biological N-fixation through symbiotic association with N_2-fixing bacteria (Carlsson and Huss-Danell, 2003; Peoples et al., 2019). Inclusion of legumes in leys can, therefore, be used to reduce dependence on synthetic N fertiliser (Ledgard and Steele, 1992), a highly resource-intensive agricultural input that causes environmental impacts from its production and use (Galloway et al., 2003; IEA, 2021; Jensen et al., 2012; Tian et al., 2019). However, diversification measures to promote ecosystem services may have differing effects depending on where and how they are adopted, and they can reduce obtainable yield (Tamburini et al., 2020). In addition, economic forces tend to favour cost-efficient specialist cropping systems over the more long-term benefits of diversification (Reckling et al., 2016; Zander et al., 2016).

Since the beginning of agriculture, soils have been a source of atmospheric carbon dioxide (CO_2) through depletion of soil organic carbon (SOC) (Lal, 2010). The rate of depletion has been accelerated by the specialisation of arable agriculture, with systems dominated by annual crops. Temporary leys have the potential to sequester considerable amounts of C in agricultural soils by shifting the SOC equilibrium to a higher level (Börjesson et al., 2018; Englund et al., 2023; Poeplau et al., 2015). High plant diversity within the ley mixture itself may also be a driver of SOC sequestration, by promoting belowground SOC input and an increased contribution from microbial necromass (Bai and Cotrufo, 2022; Kagiya et al., 2019). This SOC sequestration potential, in combination with low associated costs, has generated interest in using agricultural soils as a negative emissions approach to remove CO_2 from the atmosphere with the aim of reducing global warming (Minx et al., 2018; Smith et al., 2016). Within the European Union (EU) alone, SOC sequestration potential is estimated to be between 9 (Frank et al., 2015) and 58 million ton CO_2 per year (Lugato et al., 2014). However, SOC sequestration rate is highly dependent on, for example, soil properties, climate, farming system and current soil C content (Bolinder et al., 2020; Kätterer et al., 2012). This makes it difficult to predict the soil C effect of various cropping systems. Studies investigating soil C effects within agricultural systems often rely on modelling because of the protracted nature of soil C changes and lack of available measured data (Goglio et al., 2015; Nilsson et al., 2020a, 2020b; Poeplau et al., 2015). However, the use of models entails significant uncertainty, and the underlying theory has been challenged (Dungait et al., 2012; Schmidt et al., 2011). This uncertainty reduces the overall effectiveness of models in accounting for soil C changes (Stockmann et al., 2013), leading Goglio et al. (2015) to conclude that field data should be used where possible.

Many of the reported benefits of ley cultivation are based on field studies involving livestock, where manure is used as fertiliser on cultivated leys (Bolinder et al., 2010; Jarvis et al., 2017; Poeplau et al., 2015). This set-up makes it difficult to assess the effect of the ley itself and does not distinguish the effect of N-fixing legumes. In fact, there are currently few long-term field experiments where the effects of ley and manure on crop yields and soil C can be separated (De Los Rios et al., 2022). However, a long-term field experiment is being conducted at three sites in Sweden where crop rotations with and without perennial leys are being compared and where only mineral fertilisers have been used since the early 1970s (Persson et al., 2008).

It is important to apply a systems perspective when evaluating the environmental impacts of crop cultivation (Henryson et al., 2019). Life Cycle Assessment (LCA) is frequently used to assess the environmental impact of agricultural products and is accepted by policymakers in both public and private organisations (Brandão et al., 2022). In LCA, emissions and resources used during the whole (cradle-to-grave) or parts (e.g. cradle-to-gate) of the life cycle of a product or process are considered (ISO, 2006a, 2006b).

The overall aim of this study was to assess the effect on crop yield and environmental performance of including ley in crop rotations, focusing on the comparison between a pure grass ley and a legume-grass ley mixture at the same levels of N fertiliser application. The analysis was based on data from the long-term field experiment running at three sites in Sweden. Specific objectives were to:

- Quantify the effect of ley in crop rotations on annual crop yield in the rotation, total crop rotation yield, and SOC stock under different fertiliser regimes.
- Compare climate impact, energy resource depletion and eutrophication potential of including grass or grass/legume ley in crop rotations, using LCA methodology.

2. Methods and materials

2.1. Experimental sites and set-up

The study was based on data from the ongoing long-term field experiment at three sites in southern Sweden with different climates and soil properties: Säby (59°49 N; 17°42 E), Lanna (58°20 N; 13°07 E) and Stenstugu (57°36 N/; 8°26 E). The characteristics of these sites, which have been in operation since 1969, 1965 and 1968, respectively, are shown in Table S1. in Supplementary Material (SM). The aim of the experiment is to investigate the long-term effects of including ley in three crop rotations (*Mixed-Ley*, *Grass-Ley*, *No-Ley*) under four different N fertiliser regimes. The three crop rotations consist of six-year rotations with the first four crops in each rotation being identical and the last two being different (Table 1). Data for two of the four N fertiliser levels (the highest and second lowest, referred to here as *High N* and *Low N*) were used in this study, because SOC was only measured in these treatments. Thus in total, data from six treatment combinations (three crop rotations × two N fertiliser levels) at each site were included in the analysis.

At Säby and Lanna, the experiment follows a split-split-plot design with crop rotations and N-levels included as subplots and sub-subplots,

Table 1
The composition of the crop rotations in a long-term Swedish field experiment at three different sites in southern Sweden, which supplied data used in this study to evaluate the environmental effect of rotational leys.

Rotation	Mixed-Ley	Grass-Ley	No-Ley
I	Oilseed crop	Oilseed crop	Oilseed crop
II	Winter wheat	Winter wheat	Winter wheat
III	Oats	Oats	Oats
IV	Barley	Barley	Barley
V	Legume-Grass Ley I	Grass Ley I	Spring wheat
VI	Legume-Grass Ley II	Grass Ley II	Fallow

respectively. At Stenstugu, a split-strip-plot design is used, with crop rotations and N-levels arranged as rows and columns (Fig. S1 in SM). All six crops in each rotation are cultivated each year in neighbouring main plots. Thus, there are as many replicates as there are crops in the rotation. This design means that comparisons between the treatments can only be made over time, because each main plot is in a different position in the rotation. The Mixed-Ley treatment consists of red clover (at Säby and Stenstugu lucerne is also included in the seed mixture) and timothy, while the pure Grass-Ley is a mixture of the grass species meadow fescue and timothy. More information on the study sites and experimental set-up can be found in Persson et al. (2008).

Yield data for each treatment at Säby in the period 1969–2016, Lanna in the period 1965–2012 and Stenstugu in the period 1968–2015 were used to calculate mean yield for the entire crop rotation, and for each crop in the rotation. Mean yield was then used to compare land occupation, i.e. yield per m^2 agricultural land. Soil organic C was measured in the topsoil (0–20 cm) once per rotation (after the oat crop), except between the years 1993 and 2005. Subsoil samples (40–60 cm) were also collected and analysed. However, as no significant changes were observed during the assessed period for any of the treatments, the subsoil data was not incorporated into the LCA.

To estimate mean SOC stock change per rotation, a random intercept and slope model that takes into account the SOC change for each plot, and then calculates mean SOC change for each site and treatment (Zuur et al., 2009), was applied to the collected data (see regression plots in (see regression plots in Fig. S2, S3, and S4 in SM). The data used to assess the change in SOC were collected from the beginning of the field experiment until the most recent samples analysed in 2020. The C content (%) was converted to kg C per ha using the equation:

$$SOC \quad \left(\frac{kg \quad C}{ha}\right) = \frac{SOC(\%)}{100} \bullet \rho \bullet V \qquad (1)$$

where SOC is soil organic carbon content, ρ is soil bulk density at each site and V is volume of 1 ha of topsoil (to 20 cm depth). Using the pedotransfer functions for Swedish agricultural soils developed by Kätterer et al. (2006), topsoil density was estimated to be 1.27, 1.31 and 1.52 g cm^{-3} at Säby, Lanna and Stenstugu, respectively.

2.2. Life Cycle Assessment

2.2.1. Goal and scope

LCA methodology was used to quantify and compare the environmental impact, in terms of greenhouse gas (GHG) emissions, energy resource depletion and eutrophication potential. The system boundaries were set from cradle to farm-gate, including the complete crop rotation for *Mixed-Ley*, *Grass-Ley* and *No-Ley* (Fig. 1). Life cycle inventory (LCI) was performed for the following processes:

- Production of fertiliser and pesticides
- Seed cultivation
- Field operations, including fuel production and consumption, and production and maintenance of machinery
- Crop drying,
- Emissions to water (N and phosphorus (P) leaching) and emissions to the atmosphere (nitrous oxide (N_2O), ammonia (NH_3) and nitrogen oxides (NOx))
- SOC changes

An important concept in LCA methodology is the functional unit, which is used as the basis for quantification, i.e. the environmental impact is quantified per functional unit. The functional unit should be chosen with respect to the goals of the study, but it is sometimes not obvious which is most suitable and several can be included in the assessment (Klöpffer and Grahl, 2014). Here, we applied two separate functional units: i) kg harvested cereal units (CU) and ii) ha of agricultural land. The CU concept, which was developed by the German authorities to make agricultural productivity more comparable, converts harvested mass to CU by determining the animal feeding value of each agricultural product and normalising it to the reference crop (barley) (Brankatschk and Finkbeiner, 2014). The animal feeding value is based on the protein, lipid, fibre and carbohydrate content of the crop and the proportion fed to specific animal species (cattle, pigs, poultry and horses) (Brankatschk and Finkbeiner, 2014). The CU can be used in LCA studies to allocate environmental burden between crops in a rotation (Brankatschk and Finkbeiner, 2015; Goglio et al., 2018) and as a functional unit for the entire crop rotation (Henryson et al., 2019; Prechsl et al., 2017).

Assessments were performed for eight full six-year rotations, i.e. in total 48 years. Field operations included for the different crops were based on the average treatment for each site and crop in the study period (Table 2). According to the field experiment design, Legume-Grass Ley I in *Mixed-Ley* was only fertilised once a year, while Grass Ley I in *Grass-Ley* was fertilised twice, before and after the first cut. The second-year leys (Legume-Grass Ley II, Grass Ley II) only had one cut, to allow for oilseed crop seeding time. Application of N fertiliser in *Mixed-Ley* was decided depending on the legume proportion, with a higher percentage of legumes resulting in a lower amount of N fertiliser (or no N fertiliser if

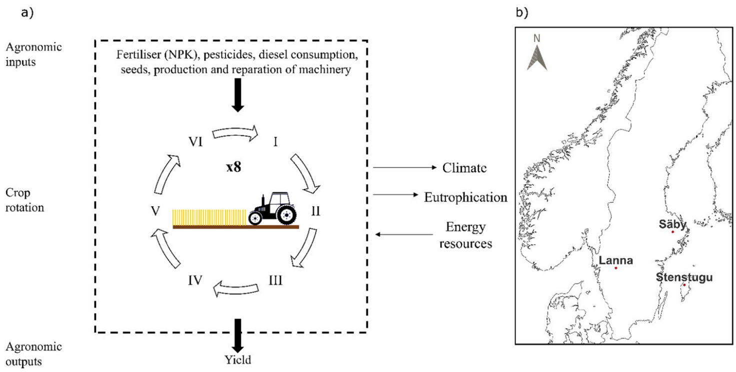

Fig. 1. a) Schematic overview of the system analysed, with assessments performed for eight full six-year rotations, and b) location of the study sites Säby (59°49´N/17°42´E), Lanna (58°20´N/13°07´E) and Stenstugu (57°36´N/18°26´E) in southern Sweden; the background map was generated using the free and open-source software QGIS.

Table 2
Field operations performed in each crop in the Mixed-Ley, Grass-Ley and No-Ley rotations in the long-term field experiment at three sites in southern Sweden.

Operation	Oilseed	Winter wheat	Oats	Barley	Mixed ley I[a]	Mixed ley II[a]	Grass ley I[b]	Grass ley II[b]	Spring wheat	Fallow
Harrowing	3	3	3	3	0	0	0	0	3	0
Fertilisation	1	2	1	1	1	1	2	1	1	0
Sowing	1	1	1	1	1	0	1	0	1	0
Application of pesticides	1	2	1	1	0	0	0	0	1	0
Harvesting	1	1	1	1	0	0	0	0	1	0
Stubble cultivation	1	1	1	1	0	1	0	1	1	0
Mowing	0	0	0	0	2	1	2	1	0	0
Ploughing	1	1	1	1	0	1	0	1	1	1

a) Only in the *Mixed-Ley* rotation. b) Only in the *Grass-Ley* rotation. c) Only in the *No-Ley* rotation.

the legume content was above 50%), based on the assumption that a high biomass would be produced with less N fertiliser if the proportion of legume was high. The mean N application over the evaluated period was utilised in LCA (Table 3). The Fallow in *No-Ley* was left untreated in the autumn after spring wheat harvesting.

2.2.2. Life cycle inventory

Data on yields, SOC content, fertiliser rates and field operations used in LCI were taken from the database for the long-term field experiment. The CU for each crop rotation and site was calculated as:

$$CU = \sum_{i=1}^{n=6} \overline{y}_i \bullet x_i \tag{2}$$

where CU (kg) is total cereal units of the crop rotation at a specific site, $\overline{y}_i$ (kg) is mean yield of crop *i* in all years assessed and x_i is the CU conversion factor for crop *i* (taken from Supplementary Material to Brankatschk and Finkbeiner, 2014) (see Table S2 in SM).

No environmental burden was allocated to the straw, since it was left on the field and was not considered an output from the system assessed. Land occupation was determined by calculating the area required to produce 1 kg CU in each of the treatment combinations.

Data on application rates of N, P and potassium (K) were taken from the field experiment guidelines (Table 3). Since the N application rate in *Mixed-Ley* varied depending on the legume content, the mean N application rate for each site and fertiliser scheme was calculated separately for Legume-Grass Ley I and II.

Emissions and energy use from production of fertiliser and pesticides, and production, maintenance and use of machinery were calculated using data from Ecoinvent (www.ecoinvent.org) (Table 4). Inputs

Table 3
Fertiliser application rate (kg nitrogen/phosphorus/potassium ha^{-1}) in the Low N and High N treatments in the long-term field experiment at three sites in southern Sweden. The values are based on experiment instructions.

Crop	Low N	High N
Oilseed	90/75/143	210/75/143
Winter wheat	45/0/0	135/0/0
Oats	40/0/0	120/0/0
Barley	*/75/293	*/75/293
Legume-Grass Ley I	* */0/0	* */0/0
Legume-Grass Ley II	* */0/0	* */0/0
Grass Ley I	80/0/0	240/0/0
Grass Ley II	45/0/0	135/0/0
Spring wheat	45/0/0	135/0/0
Fallow	0/0/0	0/0/0

* Barley was fertilised with 60 kg N ha^{-1} in both *Low N* and *High N* in the rotations with ley (where the ley was undersown in the barley), while in the *No-Ley* rotation the barley received 40 kg N ha^{-1} and 120 kg N ha^{-1} in the *Low N* and *High N* scenario, respectively. * *Amount of N fertiliser in Legume-Grass Ley was based on the legume content. In the *Low N* regime at Säby, Lanna and Stenstugu, mean N application rate was 32, 34 and 29 kg ha^{-1} in Legume-Grass Ley I and 32, 36 and 37 in Legume-Grass Ley II, respectively. In the *High N* regime, it was 89, 101 and 87 in Legume-Grass Ley I and 89, 107 and 112 in Legume-Grass Ley II at Säby, Lanna and Stenstugu, respectively.

Table 4
Data used in Life Cycle Inventory. Ecoinvent data are based on v.3.9 cut-off by classification methodology. Abbreviations denote the geographical resolution of the dataset, where RER = Europe, CH = Switzerland, SE = Sweden and GLO = Global.

Input	LCI dataset
N fertiliser*	Ammonium nitrate production, RER
P fertiliser* *	Triple superphosphate production, RER
K fertiliser* * *	Potassium chloride production, RER,
Pesticides	Pesticide production, unspecified, RER
Ploughing	Tillage, ploughing, CH
Harrowing	Tillage, harrowing, by spring tine harrow, CH
Fertilisation	Fertilising, by broadcaster, CH
Sowing	Sowing, CH
Pesticide application	Application of plant protection product, by field sprayer, CH
Harvesting	Combine harvesting, CH
Stubble cultivation	Tillage, cultivating, chiselling, CH
Mowing	Mowing, by rotary mower, CH
Heavy fuel oil	Heavy fuel oil, burned in refinery furnace, Europe without Switzerland
Electricity	Market for electricity, high voltage, SE
Diesel production	Market for diesel, Europe without Switzerland
Diesel combustion	Diesel, burned in agricultural machinery, GLO

* 33.5% N. * *20% P. * * * 47% K.

of pesticides (herbicides, fungicides and insecticides) were based on national statistics for the specific region and for specific crops for each field (SCB, 2011). The cereals in the crop rotations were assumed to be harvested at 20% dry matter (DM) and then dried to 14%, and the oilseed crop was assumed to be harvested at 15% DM and dried to 9%, based on data from Edström et al. (2005). The demand for heating oil was set to 5.4 MJ per kg of evaporated water and electricity use in the process to 17 kWh per kg grain (Edström et al., 2005). No further processing of the ley biomass was included. To account for seed cultivation, we subtracted the seed rate (6 kg seed per ha for oilseed, 210 kg for winter wheat, 205 kg for oats, 170 kg for barley and 230 kg for spring wheat) from the yield (Ahlgren et al., 2011). The seed rate for the ley crops was set at 24 kg per ha in both *Mixed-Ley* and *Grass-Ley*. Diesel use for producing the ley seeds was assumed to be 19.4 MJ per kg (Prade et al., 2015) and emissions from sowing were based on the Ecoinvent dataset (Table 4).

The calculated mean SOC change (kg ha^{-1}) was used in the LCA model to estimate the average change in SOC per rotation and was converted to CO_2 based on the atomic weight ratio of C to CO_2. Direct soil N_2O emissions were estimated using the IPCC Tier I approach (IPCC, 2019), with the emissions factor for temperate wet climates (0.016 kg N_2O-N kg N^{-1}) and mean change in SOC content per rotation. Indirect N_2O emissions were calculated using the IPCC Tier I approach, where N_2O from volatilised N and N from leaching are both included. Nitrogen leaching was estimated using the farm management tool VERA, described in Aronsson and Torstensson (2004). Phosphorus leaching was estimated using data from Johnsson et al. (2016), who calculated mean leaching rates for 22 regions in Sweden using the ICECREAMDB model. The data used represented leaching and runoff rates for specific crops and soil textures (Johnsson et al., 2016). Emissions factor for N volatilisation at field level was set to 0.033 kg NH_3 and 0.04 kg NO_x per kg applied N fertiliser (EMEP/EEA, 2016).

2.2.3. Life cycle impact assessment

The environmental aspects considered were climate impact, energy resource depletion and eutrophication potential. The climate impact was assessed using GWP_{100}, applying the characterisation factors in Forster et al. (2021). Resource use in terms of energy resources was calculated using the abiotic depletion potential method for energy carriers developed by Van Oers et al. (2002) and updated in Van Oers and Guinée (2016). This method is included in the set of indicators used in the EU Environmental Footprint version 3.0 (Crenna et al., 2019). Eutrophication potential was assessed using the CML method (Guinée, 2002), a simple approach for assessing potential eutrophication which assumes that all N and P discharged to the environment can cause eutrophication by placing the indicator at the point of emission, and thereby not including the fate of the emissions (Henryson et al., 2019).

3. Results

3.1. Yields

The yield effect of ley on the first four crops in the rotation was assessed by comparing the difference in yield for each individual crop between the ley rotations and *No-Ley* (Figs. 2a-2c). Under the *Low N* regime, inclusion of ley in the rotations at all sites gave higher mean yield of the first four crops except for oilseed, with higher yields observed in *No-Ley* than in *Grass-Ley*. The largest positive yield effect was observed for *Mixed-Ley*, which was most evident when comparing the yield expressed in CU. Under the *High N* regime, the difference between the ley rotations and the *No-Ley* rotation was small, but barley yields were considerably lower in the ley rotations at all sites, most likely because less N was applied to the barley in the ley rotations in order to ensure good establishment of the undersown ley crop (Table 3). Across all study sites, the mean aggregated effect of ley on yield of the first four crops under the *Low N* fertiliser regime was 1.69 and 0.51 Mg CU ha^{-1} for *Mixed-Ley* and *Grass-Ley*, respectively. Under the *High N* regime, the yield response was − 0.29 and − 0.45 for *Mixed-Ley* and *Grass-Ley*, respectively. Mean yield of each crop in the different crop rotation and sites is shown in Table S3 in SM.

When expressed in CU, total yield of all crops in the rotation was clearly higher in the ley rotations than in *No-Ley* (Fig. 3a). Moreover, under the *Low N* regime, total yield was higher in *Mixed-Ley* than in the *Grass-Ley* rotation, particularly at Lanna and Stenstugu. Under the *High N* regime, the opposite effect was found in Säby and Lanna, i.e. higher total yield in *Grass-Ley* compared with *Mixed-Ley*, although the difference was small. Total crop rotation yield was higher under *High N* than *Low N*. Higher total yield means lower land occupation in terms of area required to produce 1 kg CU. Consequently, the land occupation was lowest for *Mixed-Ley* under the *Low N* regime and *Grass-ley* under *High N*, respectively. In general, the *High N* regime resulted in a lower land occupation than *Low N*. Across all study sites, the mean ley yield effect on the total crop rotation in *Mixed-Ley* and *Grass-Ley* was, respectively, 6.58 and 4.23 Mg CU ha^{-1} for *Low N* and 3.72 and 4.51 Mg CU ha^{-1} for *High N*.

The contribution of each crop to total CU of the rotation was similar for the two ley rotations, where Ley I (i.e. the first year of ley) and Ley II (i.e. the second year of ley) contributed between 35% and 38% of total CU, and Ley I made a larger contribution than Ley II in both *Mixed-Ley* and *Grass-Ley* (Fig. 3b). In the *No-Ley* rotation, the largest contribution to the CU was made by the winter wheat crop, which alone accounted for 29% and 27% of total CU in the *Low N* and *High N* fertiliser regime, respectively.

3.2. Soil organic carbon

Estimated mean SOC change, which was used in the LCA, indicated that all treatments resulted in depletion of SOC, leading to atmospheric CO_2 emissions. However, there was large variation between replicate plots, as indicated by the error bars in Fig. 4. Changes in SOC in all plots are shown in Fig. S2. Under the *Low N* regime at Säby and Stenstugu, the greatest depletion of SOC stock occurred in the *No-Ley* rotation (153 and 199 kg C ha^{-1} year^{-1}, respectively) and the least depletion in the ley rotations (70 and 133 kg C ha^{-1} year^{-1} in *Mixed-Ley* and 76 and 169 kg C ha^{-1} year^{-1} in *Grass-Ley* at Säby and Stenstugu, respectively). The *High N* regime resulted in lower SOC stock depletion at Säby, and particularly at Stenstugu. However, at Lanna, there was almost no difference in SOC change between the rotations under the *Low N* regime, whereas under *High N* the greatest SOC stock depletion was found in the *Mixed-Ley* rotation (Fig. 4).

The mean difference in SOC between the ley rotations and *No-Ley* after eight rotations, i.e. 48 years, over all sites was 2.54 and 2.49 Mg ha^{-1} for *Mixed-Ley* and *Grass-Ley,* respectively, under the *Low N* regime. Under the *High N* regime, the difference was 1.43 and 2.71 for *Mixed-Ley* and *Grass-Ley*, respectively.

3.3. Life Cycle Assessment

3.3.1. Greenhouse gas emissions

The lowest GHG emissions per kg CU were found in the *Mixed-Ley* rotation under the *Low N* fertiliser regime (Figs. 5a-5c). This was most evident at Lanna and Stenstugu, where GHG emissions from *Mixed-Ley* under the *Low N* regime corresponded to 81% and 77% of those from *Grass-Ley*. At Säby, the same treatment corresponded to 90% of those from *Grass-Ley*. The highest estimated emissions per CU were found for the *No-Ley* rotation at all sites. Under the *High N* regime, the difference was smaller between the two ley rotations. At Lanna, GHG emissions per CU were lower in *Grass-Ley* than in *Mixed-Ley*. The *High N* application regime resulted in greater emissions from production of N fertiliser and

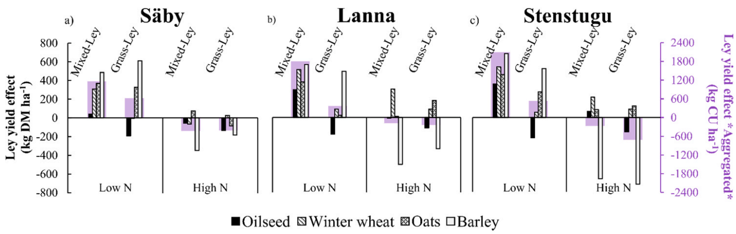

Fig. 2. Effects of ley inclusion on yield of the first four crops (oilseed, winter wheat, oats, barley) in the Mixed-Ley and Grass-Ley compared to the No-Ley rotation under the different N fertiliser regimes assessed at (a) Säby, (b) Lanna and (c) Stenstugu. Purple bars indicate difference in aggregated cereal units (CU) of the first four crops.

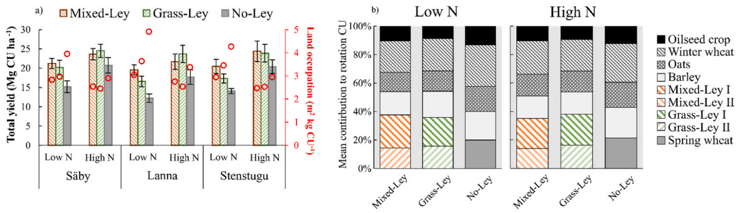

Fig. 3. (a) Mean total yield of the full rotation (cereal units (CU) ha^{-1}, error bars represent 95% confidence interval) at each site and for each crop rotation and fertiliser regime, where red circles represent land occupation (m^2 needed to produce 1 kg CU). (b) Mean contribution of each crop to total CU for each treatment.

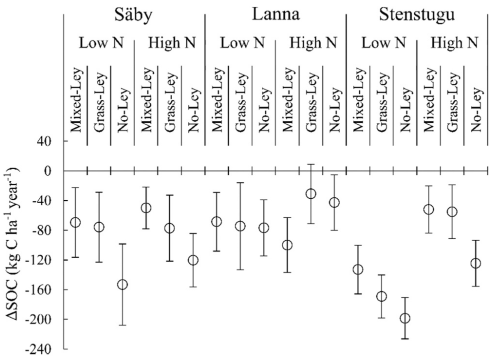

Fig. 4. Change in calculated soil organic carbon (SOC) stock in the topsoil (20 cm depth) in each treatment at the Säby, Lanna and Stenstugu sites. The mean value for each treatment is marked with a circle, error bars represent 95% confidence interval.

also greater soil N_2O emissions per ha and per kg CU. Soil organic C depletion in all treatment combinations and at all sites resulted in additional CO_2 emissions from the systems. Compared with the *No-Ley* rotation, the average GHG emissions from *Mixed-Ley* were 329 and 188 g CO_2-eq lower per kg CU for *Low N* and *High N*, respectively, whereas *Grass-Ley* resulted in 200 and 147 g CO_2-eq lower GHG emissions per kg CU for *Low N* and *High N*, respectively (Fig. 5d).

3.3.2. Energy resource depletion and eutrophication potential

Similarly to the findings for climate impact, energy resource depletion and potential eutrophication were lowest per CU in the *Mixed-Ley* rotation under the *Low N* regime (Fig. 6). In the *Low N* regime, the majority of total energy depletion originated from field operations, while in *High N* a higher proportion of energy depletion came from the agricultural inputs. This was because of the higher N fertiliser rate, which caused greater total depletion both per ha and per CU compared with *Low N*. The greatest energy resource depletion per CU was in the *No-Ley* rotation, while the greatest energy depletion per ha was in the *Grass-Ley* rotation, due to the larger total N input in that rotation.

Nitrogen emissions, predominantly in terms of leaching, contributed most to eutrophication potential at each site. Phosphorus leaching also had a considerable impact, particularly at Lanna (Fig. 6). Simulated impacts were lower for the other sources of eutrophication included in

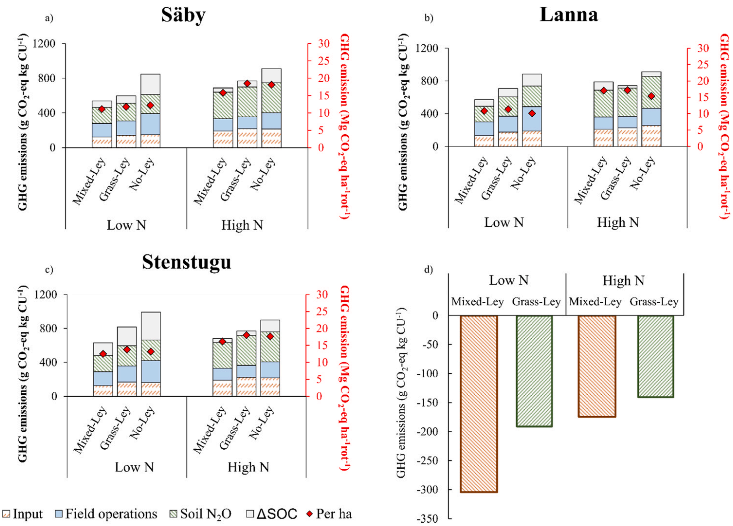

Fig. 5. Greenhouse gas (GHG) emissions per kg cereal units (CU) (bars, left axis) and per hectare agricultural land (red diamonds, right axis) at (a) Säby, (b) Lanna and (c) Stenstugu and (d) mean difference in emissions across all sites between the ley rotations (Mixed-Ley, Grass-Ley) and the No-Ley rotation (values on bars indicate total GHG emissions in g CO_2.eq per kg CU).

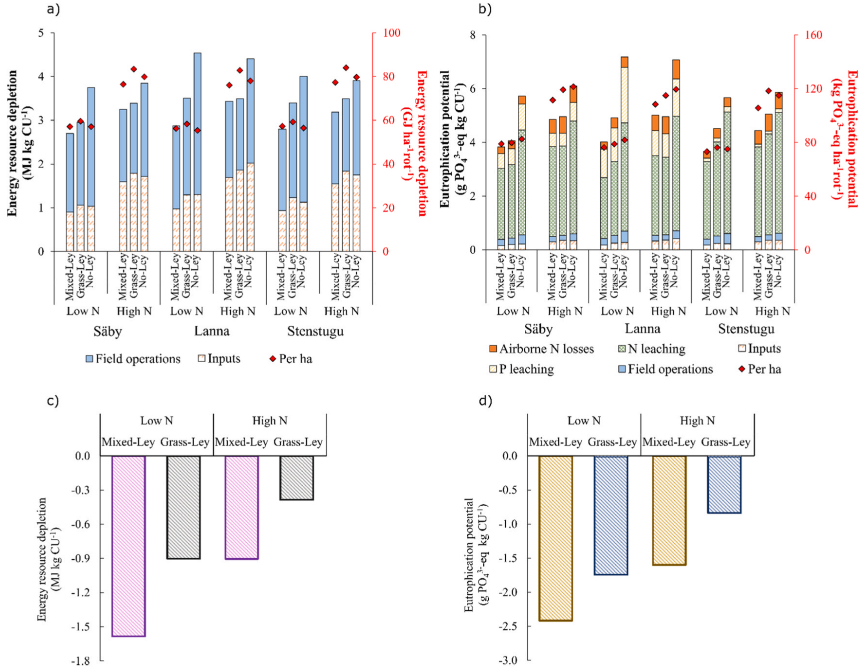

Fig. 6. (a) Fossil energy resource depletion and (b) eutrophication potential in the different treatments at the Säby, Lanna and Stenstugu sites, per kg cereal units (CU). Red diamonds indicate total impact per hectare of agricultural land for a whole rotation, and mean difference across all sites between ley rotations and the No-Ley rotation in (c) energy resource depletion and (d) eutrophication potential (values on bars indicate total impact per kg CU).

the assessment. Eutrophication potential was generally higher under the *High N* fertiliser regime, per CU and per ha agricultural land, than under the *Low N* fertiliser regime.

4. Discussion

4.1. Effects on biomass yield and soil organic carbon

Inclusion of two-year leys had a positive effect on yield of the first four crops in the rotation under the *Low N* fertiliser regime (Figs. 2a-2c). This effect was especially evident in the *Mixed-Ley* rotation (legume-grass ley), where the first four crops at all sites showed higher yields than those in the rotation without ley (*No-Ley*). A similar positive influence on yield following diversification of cropping systems has been reported in previous studies (e.g. Nunes et al., 2018; Ponisio et al., 2015). Marini et al. (2020) found specifically that diversification by including two- to three-year grass-legume leys in six- to seven-year crop rotations increased mean yield of winter and spring cereals by 0.86 and 0.39 Mg per ha and year, respectively. In the present study, aggregated mean yield of the four first crops increased by 1.69 and 0.51 Mg CU per ha for *Mixed-Ley* and *Grass-Ley*, respectively, under the *Low-N* regime. This yield effect of ley could be attributable to enhancement of ecosystem services leading to e.g. improved soil health and pest control (Tamburini et al., 2020), nutrient conservation by leys and symbiotic atmospheric N fixation in the *Mixed-Ley* rotation (MacLaren et al., 2022). Angus et al. (2015) found that the more unrelated the preceding crop, the greater the yield effect on wheat crops, which supports the theory that diversification has a positive effect on yields. In contrast, Garland et al. (2021) suggest that it is not diversification itself that is most important, but rather the proportion of the year with crop cover. A similar positive yield effect of ley inclusion was not observed for the *High N* fertiliser regime, which indicates that some of the positive effects on biomass provisioning gained through diversification were lost by increased N fertiliser application. Similarly, MacLaren et al. (2022) observed that diversification with legumes increased yields under low N fertilisation, but had little to no effect under high N fertiliser rates. In line with our results, they also found that pure grass leys had positive yield effects, indicating that perennial leys provide different provisioning functions than annual crops (MacLaren et al., 2022). Thus, diversifying crop rotations by including leys, especially mixed legume-grass leys, can be a strategy to maintain yields under a lower N application regime because of environmental concerns, high prices or ambitions to be self-sufficient.

The mean yield of the entire six-year crop rotation expressed in CU was higher in the ley rotations than in the *No-Ley* rotations (Fig. 3a), resulting in lower land occupation in the ley rotations. Land occupation is an important aspect when assessing the environmental impact of agricultural systems, as clearing new agricultural land is one of the major drivers of the negative climate and biodiversity impacts of agriculture (Foley et al., 2011). In general, crop rotation yield was higher and land occupation was lower for *Mixed-Ley* compared with *Grass-Ley* under the *Low N* regime, and vice versa under the *High N* regime. The lower total yield in the *No-Ley* rotation was partly attributable to the one-year fallow with no harvested biomass. While CU increased with the ley rotations, production of annual crops decreased. Without opportunities to harness the ley biomass, the net effect could be greater land occupation, instead of a reduction. However, there are several options for increasing ley biomass utilisation. These include expanded utilisation as feed for ruminants and monogastric animals such as pigs (Zira et al., 2023), production of protein concentrates through biorefineries for high-quality feed or food applications (Santamaría-Fernández and Lübeck, 2020; Jørgensen et al., 2022), and use as a feedstock for bioenergy (Englund et al., 2023).

The calculated mean SOC stock decreased in all treatments, with or without inclusion of ley (Fig. 4). However, the values showed large variation between replicated plots, adding uncertainty to these results. At two of the sites, Säby and Stenstugu, SOC depletion was lower in rotations with ley for both fertiliser regimes. In contrast, SOC depletion at Lanna was greatest for the *Mixed-Ley* rotation under the *High N* regime. Inputs of organic matter to soils, for example, in the form of crop residues, have been shown to, under certain conditions, accelerate microbial activity and degradation of C already present in the soil (Blagodatskaya et al., 2011). These so-called priming effects are often used as an explanation when increased C inputs lead to elevated SOC decomposition (Poeplau et al., 2015). However, the mechanisms underlying this phenomenon and their interconnections are still not fully understood (Liu et al., 2020), and we have no further evidence to indicate that this mechanism was responsible for the higher SOC depletion in the *Mixed-Ley* under the *High N* regime at Lanna.

A study assessing the average SOC content over the entire duration of the Swedish long-term experiment across all sites has found significantly lower SOC stock in the crop rotation without ley (El Khosht et al., in prep). The lack of SOC sequestration seen in the present analysis, despite ley inclusion, may be due to the ley proportion being insufficient. Jarvis et al. (2017) compared the effect on soil properties of introducing different proportions of ley (1, 2, 3 and 5 years) in six-year rotations and found that the rotations with a higher proportion of ley resulted in larger C stock in the topsoil. However, the rotations with a higher ley proportion also received manure (Jarvis et al., 2017), which has been shown to have a significant effect on long-term SOC sequestration (Bolinder et al., 2020). Similarly, Zani et al. (2021) concluded that a larger proportion of temporary leys in a rotation had a positive linear correlation with SOC concentration when the ley proportion reached 30–40% of the full crop rotation. Moreover, a study by Henryson et al. (2022) investigating SOC content using national monitoring data in Sweden found higher SOC levels on beef and dairy farms than on arable and pig farms, which they attributed to differing proportions of ley and amount of applied manure in the different farming systems.

Another reason for SOC depletion in all rotations in the present study might be the initial SOC content in the soil (Kätterer et al., 2012). We do not have information about former land use, but it is plausible that the current management scheme includes e.g. fewer perennial crops than before the long-term field experiment was initiated. The straw from the annual crops was left in the field in the present study, which could have counteracted further SOC losses, although below-ground biomass is more recalcitrant and gives higher potential for SOC sequestration than above-ground residues (Kätterer et al., 2011; Menichetti et al., 2015; Rasse et al., 2005). One strategy to reduce SOC depletion may be to return part of the biomass to the soil, e.g. in the form of manure, biogas digestate or sewage sludge, which show high recalcitrance to degradation in the soil environment and may, therefore, be important in SOC stock build-up (Kätterer et al., 2011). Our results also indicated lower SOC depletion in *High N* compared with the *Low N* regime. This is in line with Kätterer et al. (2012), who concluded that there is a positive correlation between SOC storage and mineral N applied under Swedish conditions, due to increased biomass production at higher N application rates, which in turn increases the supply of organic matter to the soil. Moreover, Kirkby et al. (2014) showed that adequate availability of soil N is essential for the formation of stable soil organic matter.

4.2. Life Cycle Assessment

At all sites, the lowest cradle-to-farm-gate GHG emissions per CU were for the *Mixed-Ley* rotation under the *Low N* regime (Fig. 5). This mainly was due to higher yields and lower use of N fertiliser, which resulted in lower GHG emissions from both upstream fertiliser production and soil N_2O emissions. Energy resource depletion and eutrophication potential followed the same trend as seen for GHG emissions, with a lower impact per CU for *Mixed-Ley*, especially under the *Low N* regime (Fig. 6). Thus the *Low N* regime gave rotations with lower environmental burden per CU produced, especially in *Mixed-Ley*, mostly because of lower inputs and maintained high yields. However, lower overall biomass production in the *Low N* regime meant that more land was needed to produce the same amount of CU (Fig. 3a). This higher demand for agricultural land could lead to clearing of new land in the worst case scenario, resulting in a considerable additional environmental burden. Nevertheless, this is unlikely in Sweden because of the rather large amount of under-utilised land. According to Olofsson and Börjesson (2016) there are 88,000 ha of abandoned arable land in Sweden, while official statistics show that the area of arable land in use in Sweden decreased by 168,000 ha between 2000 and 2022 (Swedish Board of Agriculture, 2022). There is also potential for more efficient use of cultivated biomass, e.g. by people converting to a more plant-based diet in Western societies, thereby reducing pressure on existing agricultural land (Mottet et al., 2017). Reducing meat consumption has been suggested as a measure to combat global warming (Smith et al., 2019) and alleviate other environmental and human health issues (Martin and Brandão, 2017; Röös et al., 2020). Limiting the animal husbandry sector to using agricultural residues, such as ley biomass and crop residues, and grazing pastures with biodiversity value, as suggested by Karlsson (2022), would ease the pressure on agricultural land. It would also enable more extensive agricultural practices with more diversified cropping systems, which according to our results would entail lower GHG emissions per unit harvested yield. Furthermore, the expanding bio-economy, involving the replacement of fossil products with bio-based alternatives, is expected to increase demand for biomass (Popp et al., 2014). It is imperative to ensure that this increased demand is met without causing new environmental impacts.

Increasing the N fertiliser rate (from *Low N to High N*) generally increased GHG emissions per kg CU and use of N fertiliser had the strongest climate impact in the form of soil N_2O emissions, which is in line with previous findings (Goglio et al., 2015; Henryson et al., 2019). This implies that a technology transition to reduce GHG emissions from chemical N fertiliser production would have only a moderate effect on the total life-cycle GHG emissions. Therefore, to reduce the environmental impact of agriculture, conventional farmers must end their overuse of N fertiliser and learn from systems that are less reliant on chemical fertilisers (Foley et al., 2011). This will not be an easy task as it may result in lower yield per ha, with associated loss of income for farmers. It may, therefore, be argued that the need to incentivise measures that work towards closing the N cycle and low-fertiliser input systems that provide environmental benefits should make strong cases for the establishment of financial compensation schemes (Billen et al., 2021).

Many studies have reported SOC sequestration potential from including perennial crops in crop rotations (Bolinder et al., 2010; Börjesson et al., 2018; Kätterer et al., 2012). However, we observed SOC depletion for all rotations and at all sites when including two years of ley within six-year rotations. Changes in management practices that result in less SOC depletion than in a business-as-usual scenario are often considered to contribute to mitigation of global warming (Kätterer et al., 2012). In the present study, the ley rotations generally lost less C than the rotation without ley, which means that diversification through including ley crops in pure annual crop rotations had a net mitigating effect on CO_2 emissions from the soil. Such diversification will not remove current CO_2 from the atmosphere, but will reduce the future CO_2 concentration compared with business-as-usual (in our case the *No-Ley* rotation). Furthermore, SOC depletion was generally lower under the *High N* fertiliser regime, which may indicate that increased N fertilisation would be beneficial from a climate impact perspective. However, increasing N application to enable SOC sequestration would be a perilous strategy, since global warming mitigation from SOC sequestration will only continue until a new SOC equilibrium has been reached and N_2O emissions will continue to be elevated after that point, which may turn the system from a GHG sink into a GHG source (Lugato et al., 2018).

In a European Union context, lowering the environmental impact of agriculture is currently being promoted through several regulations and incentives, such as the European Green Deal (EU Commission, 2019), the Biodiversity Strategy for 2030 (EU Commission, 2020a), and the Farm to Fork Strategy (EU Commission, 2020b). Our results show that including perennial leys in crop rotations, especially leys containing legume species, can help achieve these targets by decreasing environmental impacts, with more prominent benefits under a Low N regime. Recommended fertiliser application rates fluctuate over time depending on the prices of fertilisers, cereals and other cash crops. The war in Ukraine and the subsequent heavily reduced availability of Russian natural gas on the European market led to historically high prices of N fertiliser in 2022 (World Bank, 2022). This type of market shock may increase interest in alternative sources of N fertiliser, e.g. N-fixing legume species. Provided that a market can be found for the ley biomass, inclusion of mixed legume-grass ley in crop rotations may increase the profitability of the cropping system, while also reducing the environmental impacts.

4.3. Limits, uncertainties

The study was based on empirical data from the long-term field experiment established in Sweden in the 1960 s, so the results were less affected by the inherent uncertainties often associated with modelling of agricultural systems. However, using empirical data adds other uncertainties, e.g. due to crop failure from pest attacks and extreme weather events. Some uncertainties are also associated with measuring methods, which may have caused e.g. the large variation in SOC change, showing considerable overlap of confidence intervals for the treatments assessed (Fig. 5). Moreover, due to lack of data on soil bulk density required to convert the measured SOC content (%) to SOC stock (kg C ha^{-1}), we used the pedotransfer functions developed by Kätterer et al. (2006). This approach has been used in several other studies (e.g. Börjesson and Kätterer, 2018; Hammar et al., 2017; Henryson et al., 2022), but is highly uncertain (Kätterer et al., 2006). Earlier findings have shown that fields predominately cultivated with ley crops, such as pastures, tend to have lower bulk density than fields that are annually ploughed (Tyson et al., 1990). To minimise this potential divergence between treatments, soil cores for the SOC assessment were sampled after the oats, three years after the incorporation of the ley crops. Furthermore, management practices in Swedish agriculture have changed since the beginning of the long-term field experiment, in particular for N fertiliser rates, where the *High N* regime corresponds to normal application rates today. Moreover, black fallow was common in Swedish cropping when the long-term field experiment was started, but is now less common as efficient herbicides have become more available (Kudsk and Streibig, 2003). However, increased herbicide resistance in tandem with tougher regulations may require new modes of weed control in future (Heap, 2014), which may lead to the return of fallow. In addition, winter oilseed rape should be established in early August at northern European latitudes to be sufficiently vigorous to survive winter and produce high yields. However, few crops were harvested before early August during the early years of the long-term experiment and ley crops that could be harvested after the first harvest were the best option as a preceding crop. With climate change and the development of efficient machinery, Swedish winter crops are starting to grow earlier in spring and spring sowing is earlier. In addition, earlier-maturing varieties have become available. Together, this has provided more options for preceding crops for winter oilseed rape. Thus, the differences between crop rotations may change over time. The lack of biomass harvesting in the *No-Ley* rotation means that this may not have given a fair comparison to the ley rotations, but on the other hand that rotation included one extra year of an annual cereal crop (spring wheat) with a relatively high CU conversion factor (Table S2). Adding another crop in *No-Ley* would likely have improved the results for land occupation and presumably also for life cycle environmental impact of this rotation. In addition, the emission savings from the less frequent use of field operations in the ley rotations compared to the *No-ley* could be reduced if a transition is made from fossil fuels, to power the agricultural machinery, to renewable energy.

The results of LCA studies depend on methodological choices, e.g. of functional unit and system boundary. These choices are particularly important for agricultural systems, because they generally deliver multiple functions and outputs. In agricultural LCAs, the most common functional units are dry or fresh matter mass of harvested crop, together with area of land used (Notarnicola et al., 2017). However, it has been argued that mass is a misleading functional unit because its function often varies between crops (Henryson et al., 2019). With the approach used here, the entire crop rotation was included within the system boundary, which means that no allocation between different crops in crop rotation was needed. The CU metric has been used in earlier studies, e.g. by Henryson et al. (2019) and Prechsl et al. (2017), and is used in agricultural statistics to capture the most important nutritional functions of crops (Brankatschk and Finkbeiner, 2014). One drawback of CU is that it is based on the feeding value of the agronomic outputs, although not all outputs may be used as feed. However, since the most common use of ley is as forage (Cederberg and Henriksson, 2020) and cereals in Sweden are used more for animal feed than for human consumption (Eklöf, 2014), we believe that this was a reasonable approximation. Moreover, the livestock species in Germany and Sweden are similar (FAO, 2016), justifying use of the same CU conversion factor. The CU conversion value for ley biomass was lower than for other crops in the rotation (Table S2). However, a wider utilisation area of ley biomass, e.g. enabled by processing in biorefineries, may suggest that ley biomass is potentially undervalued in the present study. Furthermore, the largest contributor to GHG emissions from ley was soil N_2O emissions, which are highly site-specific and can vary over time and under different management schemes (Butterbach-Bahl et al., 2013). Measurements of soil N_2O emissions are scarce, often resulting in LCA practitioners using the crude IPCC Tier I model, which was also the case in this study.

5. Conclusions

This study investigated the effects of including ley in crop rotations in terms of yield response, changes in SOC stock and environmental impact (climate impact, energy resource depletion and eutrophication potential). The results showed that inclusion of leys resulted in higher yields of annual crops in the same six-year rotation under a *Low N* regime, particularly for the rotation including a grass-legume ley. A weaker effect of ley inclusion on the yield of annual crops was observed under a *High N* regime. Total yield, i.e. of all crops in the rotation, was also larger for the ley rotations than for the rotation without ley, mainly due to the one-year fallow in the *No-ley* rotation.

Comparison of mean SOC changes indicated SOC stock depletion for all rotations and both fertiliser regimes at all three study sites, possibly due to high initial SOC content and/or insufficient proportion of ley in the rotation (two years of six). There were large variations in SOC changes between replicate plots, but mean SOC depletion was greater, across all sites, in the rotation without ley than in those with ley. The *High N* regime generally resulted in less SOC depletion.

The mixed ley rotation under the *Low N* regime gave the lowest climate impact, energy resource depletion, and potential eutrophication per kg CU, due to relatively high biomass yield per ha and lower input of purchased agricultural commodities (mainly N fertiliser). The latter reduced the upstream impacts from fertiliser production, and also soil N_2O emissions. Thus, inclusion of ley decreased the dependence on purchased agricultural inputs and lowered GHG emissions from the cropping system, and can therefore be used to help meet targets on reducing the environmental impact of agricultural systems. However, successful implementation will depend on market demand for the ley biomass produced, which can be generated by strengthening incentives for its use in e.g. bioenergy production and animal feed.

CRediT authorship contribution statement

Johan Nilsson: Conceptualization, Methodology, Writing – original draft **Fatima F El Khosht:** Data curation, Investigation **Göran Bergkvist:** Conceptualization, Writing – review & editing **Ingrid Öborn:** Conceptualization, Writing – review & editing **Pernilla Tidåker:** Supervision, Project administration.

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.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.eja.2023.126888.

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This thesis assessed the climate impact of grass and cover crop cultivation for biogas production in Sweden by combining life cycle assessment methodology with process-based and statistical modelling and data from field experiments. The results showed a considerable climate change mitigation potential of replacing fossil fuels with biogas from grass and cover crops. The thesis findings offer valuable insights that can be used in creating sustainable crop-based bioenergy systems.

Johan Nilsson received his postgraduate education at the Department of Energy and Technology, Swedish University of Agricultural Sciences (SLU), Uppsala. He holds a Master of Science degree in Environmental and Water Engineering from Uppsala University.

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