Time and spatial dependent climate impact of grass cultivation and grass-based biogas systems

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Abstract

One strategy to limit global warming is to phase out fossil products and replace them with bio-based alternatives. This is often referred to as transitioning from a fossil economy to a bioeconomy. In this transition, it is important to know the environmental impact of bio-based products, since it can be greater than that of the fossil products they replace. Life Cycle Assessment (LCA) is a suitable methodology for studying the impact of bio-based products, since it encompasses the whole life cycle of the product. However, LCA rarely considers spatial and temporal variations in impacts. It also rarely includes soil processes such as soil carbon balance and only roughly estimates nitrous oxide (N_2O) emissions from soil.

In this thesis, LCA was combined with the agro-ecosystem model DNDC to include these soil processes and their variations over time and space. The combined method was used to assess climate impact and eutrophication in grass production at five sites in central and southern Sweden and the climate impact and energy balance in grass-based biogas production in Uppsala municipality, Sweden. Analysis of grass cultivation with two fertilisation rates (140 and 200 kg N ha^{-1}) at different Swedish sites revealed that the higher rate gave a lower climate impact per Mg harvested biomass, but that site properties were more important than fertilisation intensity in determining the climate impact.

Analysis of grass for biogas production, which was assumed to be cultivated on fallow land, was conducted for more than 1000 regional sites with different properties in Uppsala municipality and the whole life cycle was included (cradle to grave). The results showed large variations in impact between different sites, depending on weather conditions, soil properties, transport distances *etc*. The greenhouse gas fluxes from grass cultivation with the greatest climate impact were soil N_2O emissions and emissions from fertiliser manufacture, which contributed to global warming, and changes in soil carbon balance, which generally had a climate mitigating effect. Overall, grass cultivation increased soil carbon stocks, but this effect was highly site- and time-dependent. The grass-based biogas production system reduced the climate impact significantly compared with the reference fallow-diesel-mineral fertiliser system.

The method developed in this thesis, where LCA was combined with agro-ecosystem modelling, could be used to assess the environmental impact of agricultural systems in other regions. The results could then also be used to assist policymakers in optimising agricultural land use planning for food, feed and fuel production.

Keywords: Life Cycle Assessment (LCA), soil carbon sequestration, soil N_2O emissions, DNDC model, biomethane, digestate, greenhouse gas emissions, perennial cropping systems.

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Tid- och platsberoende klimatpåverkan av vallgräsodling och vallbaserad biogas

Sammanfattning

En strategi för att begränsa den globala uppvärmningen är att fasa ut fossila produkter och ersätta dem med biobaserade alternativ. Detta benämns ofta som omställningen från en fossil ekonomi till en bioekonomi. I denna omställning är det viktigt att studera miljöpåverkan av de biobaserade produkterna, då det har framkommit exempel där den biobaserade produkten har större påverkan än det fossila alternativet. 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 inkluderas sällan markprocesser som förändring av markens kollager och uppskattning av den potenta växthusgasen N_2O görs ofta med grovt förenklade modeller.

I denna avhandling kombinerades LCA med processbaserad jordbruksmodellering för att undersöka tids- och platsberoendet av miljöpåverkan. Den utvecklade metoden användes för att studera klimatpåverkan och övergödningen från vallodling på fem olika platser i Sverige, samt klimatpåverkan och energibalansen för vallbaserad biogasproduktion i Uppsala kommun. Odlingen av gräsvall inkluderade två olika gödselgivor, 140 och 200 kg N ha^{-1}. Resultatet visade att vallodling med högre gödselintensitet hade en lägre klimatpåverkan per skördat ton biomassa. Men i det stora hela hade platsens egenskaper, i form av väder och jordtyp, större betydelse för klimatpåverkansbedömningen än gödselnivån.

Den vallbaserade biogasproduktionen antogs odlas på outnyttjad jordbruksmark i träda. Totalt omfattades över 1000 olika platser i studien, alla med olika förhållanden. I denna studie inkluderades biogasens hela livscykel, från vagga till grav, vilket innebar att även biogasens rötrest inkluderades inom systemgränsen. Resultatet visade stor variation i biogasens klimateffektivitet beroende på var vallen odlades i regionen. De största växthusgasflödena var i form av utsläpp av lustgas från marken, utsläpp från framtagning av gödsel samt förändring av markens kollager. De två första bidrog till ökad växthuseffekt, medan den sistnämna minskade systemets klimatpåverkan. Generellt innebar vallgräsodlingen ett ökat kollager i marken, men denna effekt var mycket rums- och tidsberoende. Totalt gav den vallbaserade biogasproduktionen en betydande klimatreduktion jämfört med referenssystemet. Den framtagna metoden i denna avhandling, där LCA kombinerades med process-baserad jordbruksmodellering, kan användas för att studera miljöpåverkan av jordbrukssystem i andra regioner. Dessutom kan metoden användas för att bistå beslutsfattare för att optimera användning av jordbruksmark för mat-, foder och bränsleproduktion.

Nyckelord: Livscykelanalys (LCA), markkol, lustgas, DNDC, bio-metan, digestat, växthusgaser, växtodling, övergödning

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A society grows great when old men plant trees whose shade they know they shall never sit in

Greek proverb

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. Assessing the climate impact and eutrophication of grass cultivation at five sites in Sweden. Submitted

II Nilsson, J., Sundberg, C., Tidåker, P. & Hansson, P-A. Regional variation in climate impact of grass-based biogas production: A Swedish case study. Submitted

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

I Planned the paper and developed the modelling approaches together with the co-authors, performed the modelling and analysed the data. Wrote the paper with support from the co-authors.

II Planned the paper and developed the modelling approaches together with the co-authors, performed the modelling and analysed the data. Wrote the paper with support from the co-authors.

Abbreviations

AGTP	Absolute global temperature potential
AGWP	Absolute global warming potential
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalents
DM	Dry matter
DNDC	DeNitrification DeComposition model
ER	Energy ratio
FU	Functional unit
GHG	Greenhouse gas
GTP	Global temperature potential
GWP	Global warming potential
ha	Hectare (10^4 m^2)
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
J	Joule
N_2	Di-nitrogen gas
N_2O	Nitrous oxide
NH_3	Ammonia
NH_4^+	Ammonium
NO_3^-	Nitrate
RF	Radiative forcing
SOC	Soil organic carbon
SOM	Soil organic matter

1 Introduction

Around 80% of global energy consumption is currently fossil-based (IEA, 2019). This is not sustainable, since burning fossil fuels makes a strong contribution to global warming. According to the Intergovernmental Panel on Climate Change (IPCC), human-induced warming up to 2017 increased the global mean air temperature by approximately 1 °C compared with pre-industrial levels (IPCC, 2018). Global warming has already affected people world-wide and the environment, and continued warming is projected to result in long-lasting and even irreversible impacts, such as loss of ecosystems, sea level rise and ocean acidification (IPCC, 2014).

One strategy to mitigate global warming is to phase out fossil energy sources and replace them with bio-based alternatives, a change that is often referred to as transitioning from a fossil economy to a bioeconomy. In Sweden, one of the greatest challenges to this transition lies in the transport sector, where 77% of all fuel used is fossil-based (SEA, 2019). Based on current trends, the Swedish Environmental Protection Agency (SEPA) estimates that biofuel demand from the transport sector will double by 2030, from 20 to 40 TWh (SOU, 2019).

Combustion of biofuels is often considered climate-neutral, based on the rather simplistic assumption that emissions from biofuel use are compensated for by biomass regrowth. However, biofuel production entails greenhouse gas (GHG) emissions, due to inputs throughout the production chain. Land use, in terms of feedstock cultivation, also affects the GHG balance of biofuels, for example via changes in soil carbon (C) storage and emissions of nitrous oxide (N_2O) from soil. Hence, to capture the full impact of a biofuel, the whole life cycle of the system must be analysed. This can be done using Life Cycle Assessment (LCA), a comprehensive approach that considers environmental impacts throughout the whole lifespan of the product analysed (Cherubini & Strømman, 2011).

Biogas is a competitive biofuel option, typically generated from anaerobic digestion of organic wastes. Besides energy, the co-produced digestate can be used as organic fertiliser, reducing the demand for mineral fertiliser and adding carbon to the soil. Grass crops are often suggested as suitable feedstock for biogas production (*e.g*. Smyth *et al.*, 2009; Börjesson & Tufvesson, 2011; Auburger *et al.*, 2017). One reason for this is that grass cultivation is a well-proven agricultural practice that can be implemented in a wide range of conditions, without the need for new farming practices (Smyth *et al.*, 2009). Moreover, studies have shown that perennial crops, such as grasses, are more likely to sequester soil carbon than annual crops (Bolinder *et al.*, 2010). This has been shown to be an important factor in carbon footprint calculations for bioenergy systems (*e.g*. Tidåker *et al.*, 2014; Hammar *et al.*, 2017; Yang *et al.*, 2018).

Feedstock cultivation for bioenergy production demands arable land, which is a limited resource. However, the possibility exists to expand agricultural activities using set-aside arable land with no current agricultural production. This type of land is suggested to be especially suitable for energy crop cultivation, due to low short-term competition with food production and lower environmental impact than conversion of natural land (Tilman *et al.*, 2009). However, the amount of bioenergy that could be produced using this land needs to be investigated, as do the environmental effects when the set-aside land resource is utilised at various scales.

Assessing crop-based biogas systems is often complex, because agriculture is highly affected by spatial and temporal variability in *e.g*. climate, soil properties and transport distances. This means that the environmental impact can vary substantially depending on where the cultivation takes place (Henryson *et al.*, 2019). Despite this spatial and temporal dependency, LCA studies that include fine-scale spatial differentiation over time and space are quite rare, due to the large data requirement (Nitschelm *et al.*, 2016). Moreover, soil processes have repeatedly been excluded from LCA studies (Brandão *et al.*, 2011). Advances in life cycle impact assessment (LCIA) methodology during recent years have increased its temporal and spatial resolution (*e.g*. Ericsson *et al.*, 2013; Henryson *et al.*, 2018). However, reliable dynamic inventory data are commonly lacking. Measurements are often time-consuming and costly and are therefore not included in the standard LCA procedure, where practitioners usually rely on databases with low temporal and spatial resolution (Rebitzer *et al.*, 2004).

In parallel with development of the LCA methodology, much effort has been devoted to developing agro-ecosystem models for investigating processes in agricultural soils and in plant production. This work has resulted in a range of different models, *e.g*. Daycent (Parton *et al.*, 1998), Daisy (Abrahamsen & Hansen, 2000) and DNDC (Li *et al.*, 1992). By using such models to fill data gaps in time-dynamic LCAs, more information could be obtained about the spatial and temporal variability in bioenergy systems.

2 Aim, objectives and structure of the thesis

The overall aim of this thesis was to obtain information about the climate impact and mitigation potential of grass cultivation and grass-based biogas systems. This was done by combining agro-ecosystem modelling with life cycle assessment methodology. Climate impacts were assessed considering spatial and temporal variations. Specific objectives were to analyse:

- The influence of spatial and temporal variations on the life cycle climate impact of grass cultivation in a Swedish context and at different nitrogen fertilisation intensities (Paper I-II).
- The life-cycle climate impact of a grass-based biogas production system, including soil processes, using site-differentiated data (Paper II).

The work performed in this thesis is depicted graphically in Figure 1. In Paper I, the climate impact and eutrophication impact of grass cultivation were investigated at five sites in southern and central Sweden. The investigation focused on the environmental effect of grass cultivation (cradle to farm-gate) and analysed the gross effect, *i.e.* no reference scenario was used. In Paper II, the investigation was expanded to include handling of the grass biomass produced, in terms of biogas production and use of the residual digestate as fertiliser (cradle to grave). That investigation 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 biogas system was assumed to replace a reference fallow land, diesel fuel and mineral fertiliser-based system.

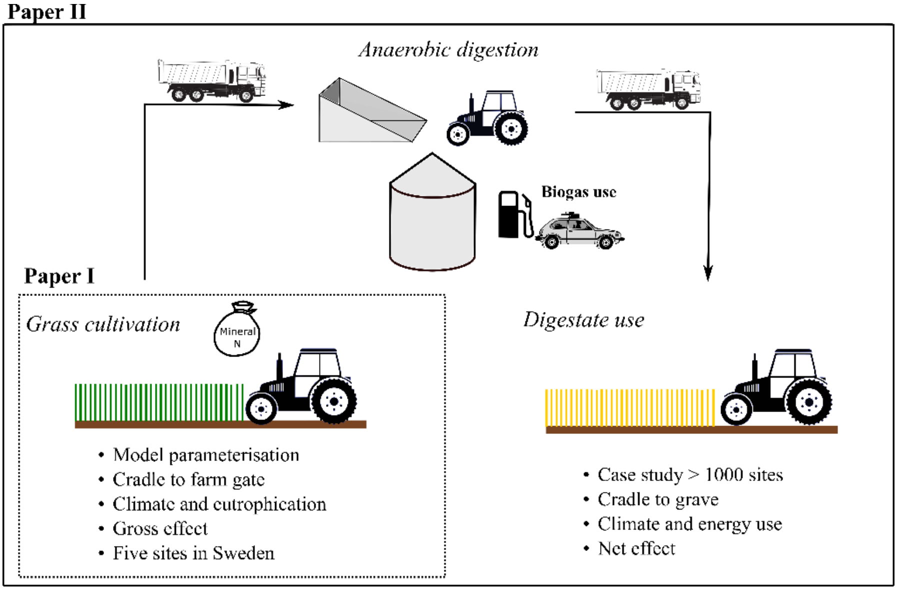

Figure 1. Schematic illustration of the work reported in Papers I and II and the links between the papers.

3 Background

3.1 Global warming and climate change mitigation

The greenhouse effect is the warming of the Earth's surface through gases that prevent infrared thermal radiation from escaping the atmosphere. These gases are often referred to as greenhouse gases (GHGs). An increased concentration of GHGs in the atmosphere leads to more outgoing thermal radiation being trapped, which causes distortion in the global energy balance. This distortion, *i.e*. the difference between ingoing and outgoing radiation, is called radiative forcing and is expressed in W m^{-2}. Increased radiative forcing leads to the so-called enhanced greenhouse effect, which means that more energy is trapped in the atmosphere resulting in an increased mean global temperature and climate change (Myhre *et al.*, 2013b), with multiple potential detrimental consequences.

Several global warming mitigation targets have been adopted worldwide to try to limit global warming. The most prominent example is the Paris Agreement signed by the member states of the United Nations Framework Convention on Climate Change (UNFCCC) at the 21st Conference of the Parties (COP 21). The Paris Agreement states that global warming is to be limited to well below 2 °C by the end of this century and that efforts to stay under 1.5 °C should be pursued (UNFCCC, 2016).

The European Union (EU) has established its own targets to battle global warming. The first milestone was the 20-20-20 target, which entailed a 20% cut in GHG emissions from 1990 levels, 20% energy production originating from renewable sources and a 20% increase in energy efficiency, all by 2020 (EU, 2016a). For the next period, 2021 to 2030, continued emission cuts are targeted, at least 40% cuts compared with 1990, at least 32% renewables and a 32.5% increase in energy efficiency (EU 2016b).

In 2017, the Swedish parliament agreed to adopt a climate policy framework, which includes a 40% cut in emissions by 2020, 63% by 2030, 75% by 2040 and no net territorial emissions by 2045. The policy framework involves a specific target for the Swedish transport sector of a 70% cut in emissions by 2030 compared with 2010 levels. The emission cut milestones referred to in the policy framework are for emissions not included in the EU emissions trading system (ETS) framework, while the 'no net territorial emissions' target comprises all emissions (SEPA, 2019). The Swedish commission tasked with evaluating progress towards the targets reported early in 2020 that the stated targets are not achievable under current policies (Climate Policy Council, 2020). Similar statements have been made regarding the Paris Agreement targets (Rogelj *et al.*, 2016; Peters *et al.*, 2017).

To make the environmental targets achievable, the world must promptly reduce emissions of GHGs from all sources, and in particular fossil sources. One strategy to achieve this is to use biofuels as an alternative to the finite, geopolitical unstable and global warming fossil fuel. Biofuels are energy-enriched chemicals generated from biomass material, such as plants, microalgae and bacteria (Rodionova *et al.*, 2017). One such biofuel, biogas (biomethane), can be used for replacement of fossil fuels in heat and power generation and in transportation. The biogas process has been used for centuries in human-made systems for energy production (Bond & Templeton, 2011). Biogas is formed from organic materials that are decomposed by a suite of microorganisms in anaerobic environments. The resulting gas distribution is dependent on the substrate, but typically consists of methane (CH_4, 50-70%), carbon dioxide (CO_2, 25-50%) and small amounts of other gases and water vapour (Plugge, 2017). Furthermore, biogas is a storable energy carrier that can be saved for future usage (Weiland, 2010). It may therefore fit well into renewable energy systems with a large share of intermittent energy sources.

Most suggested pathways for meeting the current climate mitigation targets normally comprise negative emissions technologies (NETs) (Clarke *et al.*, 2014), which remove and isolate GHGs from the atmosphere with the intention of reducing warming. NETs include technologies such as carbon capture and storage (CCS), whereby carbon is removed from the atmosphere and stored underground (Bui *et al.*, 2018). Carbon capture and storage can be combined with *e.g.* bioenergy (BECCS) or direct air capture via chemical reactions (DACCS). Other NETs include enhanced weathering on land and in oceans and ocean fertilisation (Minx *et al.*, 2018). However, the reliability in large-scale deployment of these technologies is still being debated (Anderson & Peters, 2016), and recent reviews have highlighted the lack of upscaling studies (Minx *et al.*, 2018; Nemet *et al.*, 2018).

3.2 Soil carbon and grass cultivation

One NET that has received growing interest in recent years is to increase the carbon concentration in soil through changes in agricultural practices (Smith *et al.*, 2016; Minx *et al.*, 2018). Soils store more than three times as much carbon as the atmosphere (Lal, 2004). This means that even small changes in soil carbon concentration can have a considerable effect on the global carbon balance. This is highlighted by the “4 per 1000” initiative, which was launched at COP 21 with the objective of promoting soil carbon sequestration as an important tool in climate mitigation schemes. The name of the initiative originates from the calculation that if soil carbon 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; 4per1000, 2018). Increasing soil carbon storage is not only beneficial for climate change mitigation, but also improves soil quality, for example through increased water-holding capacity, a more steady supply of nutrients, improved soil structure and reduced risk of soil compaction (Lal, 2004).

Soil carbon storage is a balance between carbon inputs, in the form of roots, crop residues *etc*., and carbon outputs in the form organic matter degradation and carbon leaching. For soils that are in equilibrium, *i.e*. where carbon inputs are equal to carbon outputs, an increase in carbon inputs will result in an increased soil carbon stock and soil carbon sequestration. The carbon stock will continue to increase until the soil reaches a new dynamic equilibrium, which can take a long time (Smith, 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 soil carbon sequestration will always have a finite climate mitigation capacity (Smith, 2014), and that the effect will vary both between different locations and between different points in time for a particular mitigating scheme (Kätterer *et al.*, 2012). Furthermore, soil carbon sequestration is a reversible process, which means that sequestered carbon can be re-emitted to the atmosphere at any time, for example if the continuity in land management is broken. Soil carbon loss typically happens faster than soil carbon build-up (Smith, 2005). These properties make soil carbon sequestration challenging to predict and handle from a policy perspective.

Earlier studies have shown that soil carbon 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 seasons (Baker *et al.*, 2007; Bolinder *et al.*, 2010; Börjesson *et al.*, 2018). One of the most commonly grown crops world-wide is grass, which in Sweden is cultivated on about 40% of all arable land (Swedish Board of Agriculture, 2018). Grass is a perennial crop, cultivated in either permanent stands or temporary leys. In temporary leys, the grass is regularly re-sown or incorporated in crop rotations (Allen *et al.*, 2011). The grass produced is typically used as fodder, but alternative uses are frequently discussed, for example in protein extraction or as feedstock in biofuel production (Tilman *et al.*, 2006; Auburger *et al.*, 2017; Carlsson *et al.*, 2017; Santamaría-Fernández *et al.*, 2017). Grass is normally grown as a mixture of species, sometimes with the inclusion of clover. The advantage of using a combination of species is that they can utilise different niches, both spatially and temporally. This means that a well-tailored mix often results in higher yields than leys of single species (Fogelfors, 2015). While grass species are dependent solely on available nitrogen in the soil, clover species can host nitrogen-fixing bacteria that provide the plant with nitrogen derived from the atmosphere. This feature makes clovers more robust and means that they can be produced in reasonable quantities with minimal energy input. However, grass species are generally more efficient at absorbing available soil nitrogen, which makes them more competitive in fertilised soils and often provides larger yields than obtained for clover species (Fogelfors, 2015).

Trials in northern Sweden have shown that including a higher frequency of perennial crops in crop rotations results in higher carbon stock than in rotations based mainly on annual crops (Bolinder *et al.*, 2010). Moreover, Swedish national inventories of agricultural mineral soils have shown that carbon stocks have increased over the past three decades, which has been attributed to an increased area of grass cultivation to support an increasing Swedish horse population (Poeplau *et al.*, 2015a). Other strategies to increase soil carbon involve recycling of organic material, use of cover crops and nitrogen fertilisation (Kätterer *et al.*, 2012).

Grass is often suggested as an energy- and climate-efficient substrate for biogas production (*e.g*. Tilman *et al.*, 2006; Smyth *et al.*, 2009; Auburger *et al.*, 2017). Anaerobic digestion of plant material is associated with some difficulties regarding process stability. However, co-digestion with other substrates, such as manure, household waste and sewage sludge, has been shown to increase the stability in the process (Nordberg *et al.*, 1997). Besides energy, the biogas production system also produces digestate that can be used as organic fertiliser, reducing the demand for mineral fertiliser and adding carbon to the soil. However, it is important to consider the emissions that occur during production of biofuel, since in some studies the bioenergy system has been shown to have a greater life cycle climate impact than the fossil-based system it was intended to replace (Creutzig *et al.*, 2015).

3.3 Environmental impact assessment

3.3.1 Life Cycle Assessment

The LCA methodology is a comprehensive approach that aims to include the impacts over the lifetime of the product investigated (cradle to grave), although the focus can be directed towards a part of the system, such as the production phase (cradle to gate). There are various ways to perform an LCA, but the globally accepted framework is regulated by the ISO LCA standard, which is essentially described in standards 14040:2006 and 14044:2006 (ISO, 2006a;

ISO, 2006b). This framework divides the assessment into four phases: (i) goal and scope definition, (ii) inventory analysis, (iii) impact assessment and (iv) interpretation. In phase (i), the intention of the LCA is formulated, *i.e*. why the study is being performed, possible target groups and whether the results are intended to be comparable to those of other products and services. In phase (ii), data on all relevant inputs and outputs required to meet the stated goal and scope are collected. In phase (iii), the inventory data collected are aggregated into specific environmental impacts such as climate impact, eutrophication, acidification *etc*. Finally, in phase (iv), the results are interpreted and put into perspective and suggestions are made for possible improvements. All four phases are performed iteratively, meaning that they can be adjusted at any time throughout the LCA process.

An important concept in the LCA methodology is the functional unit (FU), which is used as the basis for quantification, *i.e*. the environmental impact is quantified per FU. The FU should describe the function of the investigated system and can be either input-based (*e.g*. hectares of land) or output-based (*e.g*. MJ biofuel produced). The chosen FU should be described in the goal and scope phase of the LCA. It is not always obvious which FU is most suitable for the assessment, and in such cases several units can be included in the assessment (Klöpffer & Grahl, 2014).

In LCA, the most common approach for assessing the climate impact is as global warming potential (GWP) (Cherubini & Strømman, 2011). The GWP is calculated as the cumulative radiative forcing of a GHG compared with the cumulative radiative forcing of the same amount of CO_2 over a specific time horizon, typically 100 years (Myhre *et al.*, 2013b). The GWP is calculated as:

$$GWP_i(H) = \frac{AGWP_i(H)}{AGWP_{CO_2}(H)} = \frac{\int_0^H RF_i(t)dt}{\int_0^H RF_{CO_2}(t)dt} \qquad (1)$$

where AGWP is the absolute cumulative radiative forcing (RF) over a specific time horizon H. Since the emissions are relative to CO_2, the climate impact is given in CO_2-equivalents (CO_2-eq). There are pre-defined GWP characterisation factors for most GHGs. For example, the factor for CH_4 is 34 and that for N_2O is 298, with the inclusion of climate-carbon feedbacks (Myhre *et al.*, 2013a). The GWP approach does not include timing of the emissions. Instead, the emissions that occur at different points in the life cycle are added together, even though the endpoint of the impact differs (Kendall, 2012).

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. This is achieved by applying radiative forcing calculation in combination with the temperature response to changes in the radiative forcing. By investigating the cumulative absolute global temperature potential (AGTP) from the yearly emissions modelled in the life cycle inventory, the temperature response 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).

The LCA methodology was originally developed as a site-independent tool for industrial processes, but it has also been applied to other types of systems. For example, it has been used to evaluate the environmental impact of agricultural systems (Garrigues *et al.*, 2012). In contrast to industrial processes, agricultural systems contain intermediate diffuse sources with large variability both spatially and temporally. One example of this is emissions of GHGs, which are highly dependent on spatial and temporal properties, such as climate, soil type and management practices (Miller *et al.*, 2006). Furthermore, soil processes, such as soil carbon sequestration, are rarely included in LCA (Brandão *et al.*, 2011), although studies have shown that changes in soil carbon 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). Today, many environmental and administrative decisions are made on local or regional level. For this reason, it is relevant to include spatial and temporal gradients of the impact within the study area. Some previous studies have integrated spatially explicit assessment of agricultural systems (Humpenöder *et al.*, 2013; Hörtenhuber *et al.*, 2014; Henryson *et al.*, 2019). These studies highlight the importance of spatial differentiation to obtain more relevant results than those of classic LCA studies. However, introducing temporal and spatial dependency in LCA will increase the data requirement, which can cause problems for the analyst.

3.3.2 Agro-ecosystem modelling

Measurements of environmental emissions are often lacking due to high cost, time constraints and technical feasibility. The second best option is to use models. Agro-ecosystem models are used to model processes within the agricultural environment. These models are increasingly used in environmental planning and management for agriculture (Tonitto *et al.*, 2018). Agro-ecosystem models 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, 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 scientific knowledge of the modelled processes (Tonitto *et al.*, 2018). This means that the results from process-based models must be carefully scrutinised. Agro-ecosystem models have been used in multiple LCA studies to fill data gaps in the life cycle inventory (*e.g.* Bessou *et al.*, 2013; Goglio *et al.*, 2014; Kløverpris *et al.*, 2016; Deng *et al.*, 2017). Two of the most important soil processes affecting the climate impact of agricultural systems are the soil carbon balance and soil N_2O emissions.

Modelling soil carbon balance

The soil carbon 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 soil carbon 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, eventually reaching an inert pool (humus) (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, however, and therefore data on carbon input are necessary to operate them. In contrast, dynamic crop-climate 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).

Modelling soil N_2O emissions

The most important processes for the evolution 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 for example through net mineralisation of organic nitrogen, deposition from the atmosphere or via mineral fertiliser (*Figure 2*). During nitrification, N_2O is formed as a by-product to varying degrees. Under aerobic conditions, less than 1% of the oxidised NH_4^+ ends up as N_2O (Ågren & Andersson, 2012). 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*).

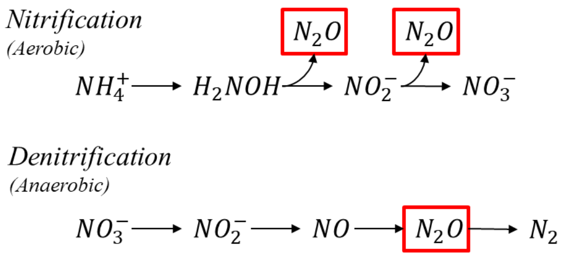

Figure 2. Production and consumption of the different reactants in nitrification and denitrification.

Nitrous oxide is a very potent climate forcer, around 298-fold stronger than CO_2 over a 100-year perspective (Myhre *et al.*, 2013a), 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). 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 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 Method

4.1 System description

In Paper I, the impact of grass cultivation was investigated at five sites spread across southern and central Sweden. In Paper II, the impact of a system ranging from grass cultivation at more than 1000 sites in Uppsala municipality, central Sweden, to biomass conversion and use of the digestate as fertiliser was studied using a life cycle approach. In Paper II, the consequence of implementing the system using existing fallow land in the region was assessed and the altered system was compared with a reference fallow land, diesel fuel and mineral fertiliser-based scenario. In contrast, in Paper I only the gross effect of grass cultivation was investigated.

4.1.1 Grass cultivation

The five sites assessed in Paper I ranged from Kungsängen in east-central Sweden to Tönnersa in the south-west (*Figure 3*).

Figure 3. Map indicating the location of the five study sites in central and southern Sweden used in Paper I.

Information about the five different sites used in Paper I is presented in Table 1. Individual site weather data for the 30-year period 1986-2015 were collected from nearby meteorological stations.

Table 1. *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
Mean annual precipitation (mm) 1986-2015	691	679	568	598	791
Soil texture	Clay loam	Sandy loam	Clay	Silty clay loam	Sandy loam
Soil organic carbon at surface (%)	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 and analysed over 30 years. The rotation started with sowing and rolling in year 1 and ended with ploughing in year 5 (*Figure 4*). During the crop rotation, the grass was assumed to be fertilised and cut twice a year. Two fertiliser intensities were investigated, 140 kg N ha^{-1} (F1) and 200 kg N ha^{-1} (F2). The first fertiliser-spreading occasion was 1 May (80/120 kg N ha^{-1}) and the second occurred after the first cut on 10 June (60/80 kg N ha^{-1}). The environmental impact was assessed per hectare (ha) of land and per Mg dry matter (DM) yield.

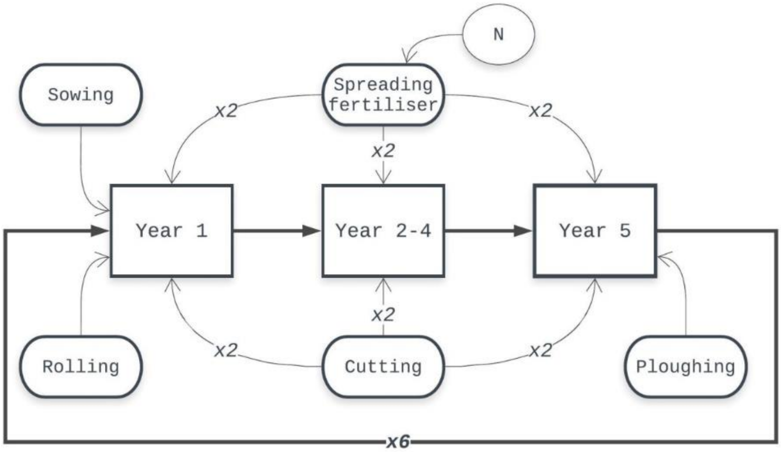

Figure 4. Schematic overview of the grass cultivation rotation analysed at the five sites in Paper I. The grass was sown and the soil was rolled in year 1 and the grass was terminated with ploughing to 30 cm in year 5. During the crop rotation, the grass was fertilised and cut twice a year.

4.1.2 Grass-based biogas

In Paper II, the system was expanded to also comprise continued handling of the harvested biomass to biogas production. The investigation was performed as a case study in Uppsala municipality, where 3587 ha were reported to be under fallow in 2014. Information about the current land use was obtained directly from the Swedish Board of Agriculture. The sites investigated (N=1240) primarily had fine-textured soils, with around 90% defined as silty clay loam, clay loam, silty clay and clay (*Figure 5*). All organic soils and fields smaller than 0.5 ha were omitted from the analysis, which reduced the total area to 3006 ha. The initial carbon content in the remaining mineral soils showed large variation, ranging between 0.7 and 11.5 %, with a median value of 2.2%. The soil pH ranged between 5.1 and 8.1, with a median value of 6.5. Data for weather conditions were obtained for 10-year period, 2007-2016. This 10-year weather sequence was looped in simulations for a 100-year period. Mean annual precipitation in the period was 596 mm and mean annual temperature was 6.5 °C.

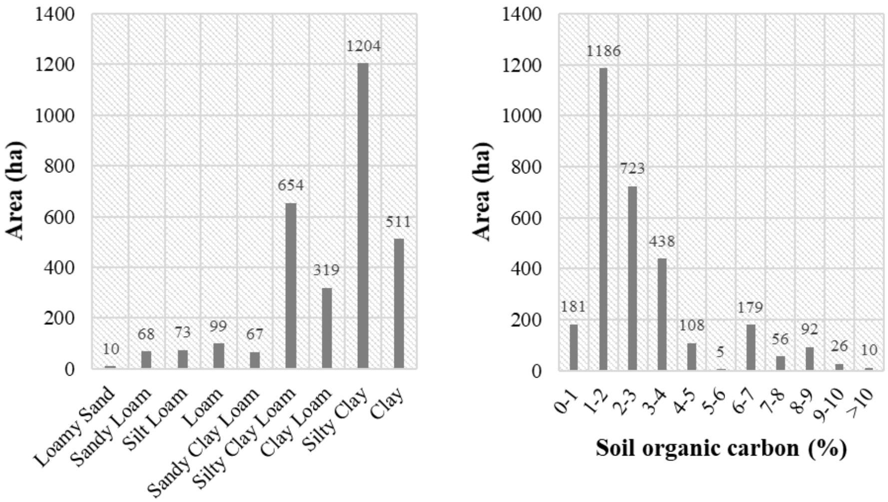

Figure 5. Soil texture and initial soil organic carbon (SOC) content at sites under fallow (N=1240) in Uppsala municipality used in Paper II.

Biogas production in the city of Uppsala is currently based on both food waste and sewage sludge and amounts to 162 TJ y^{-1}. The biogas plant in the grass-based biogas scenario in Paper II was assumed to be located at the same site as the existing municipal biogas plant (*Figure 6*). The system boundary did not include capital goods, such as construction and production of machinery. The impact of the system was assessed: (i) per ha, (ii) per MJ biogas produced and (iii) for all investigated sites in Uppsala municipality. The same grass cultivation management regime as in Paper I was assumed. The fallow land in the reference system was assumed to be left unmanaged throughout the study period except for cutting once a year in late autumn, with the cut biomass left in the field. The system was analysed over 100 years.

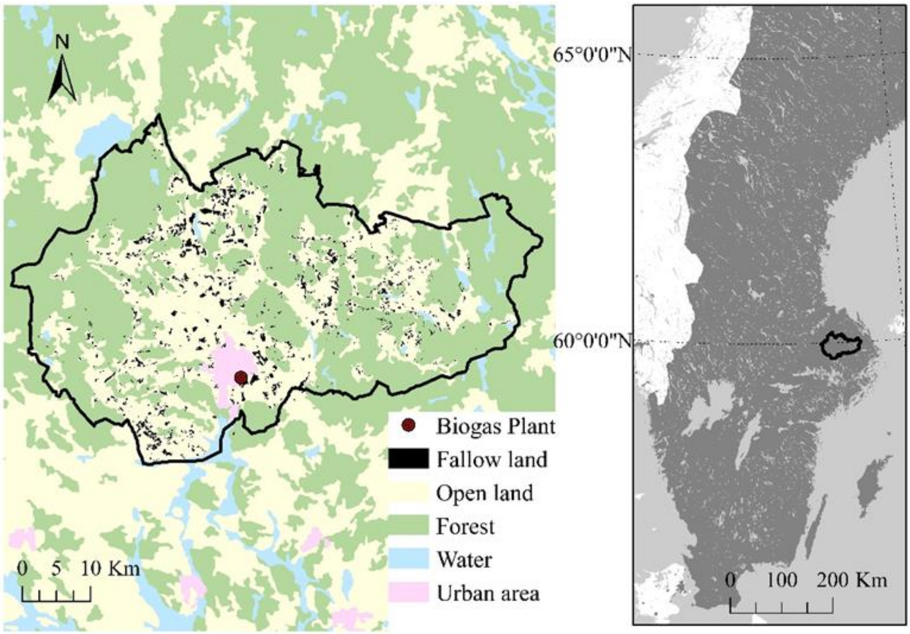

Figure 6. (Left) Map of the study region of Uppsala municipality (inside the black line), showing the distribution of fallow land (black dots) and the location of the municipal biogas plant (red and black dot). (Right) Map of Sweden showing the location of Uppsala municipality.

The grass-based biogas 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) (Figure 7). The first three subsystems comprised the altered system (A) and the latter three the reference system (R). The investigated systems were also divided into three compartments, land use (ΔLU), fuel production (ΔFP) and soil fertilisation (ΔSF). The emissions 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 comparison in the ΔLU compartment was related to the field area, *i.e*. the calculated emissions were based on the same area of grass cultivation and fallow. The comparison in the ΔFP compartment was based on engine energy, and that in the ΔSF compartment on nitrogen (N) uptake. The total GHG emissions were calculated as the difference between the altered system and the reference system as:

$$E_{Tot} = \overbrace{(E_{GrassC^A} - E_{Fall^R})}^{E_{\Delta LU}} + \overbrace{(E_{BioC^A} - E_{Foss^R})}^{E_{\Delta FP}} + \overbrace{(E_{Dig^A} - E_{Min^R})}^{E_{\Delta SF}} \quad (2)$$

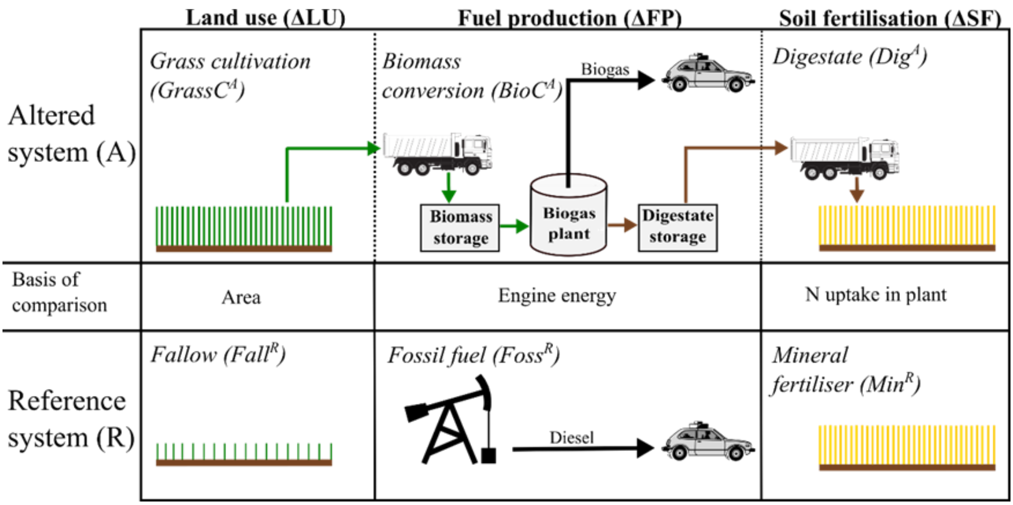

Figure 7. Schematic illustration of the grass-based biogas system studied in Paper II, divided into six subsystems: Grass cultivation (*GrassC^A*), Biomass conversion (*BioC^A*), Digestate use (*Dig^A*), Fallow (*Fall^R*), Fossil fuel (*Foss^R*) and Mineral fertiliser (*MinR*). The net effect of the system was calculated as the difference between altered system and reference system. The subsystems were also divided into three compartments: Land use (*GrassCA* - *Fall^R*), Fuel production (*BioC^A* - *Foss^R*) and Soil fertilisation (*Dig^A* - *Min^R*). The basis of comparison is shown in the row between the altered system and the reference system.

4.2 Agro-ecosystem modelling

The DNDC model was used in Papers I and II to generate data for the life cycle inventory regarding biomass yield, soil carbon changes and soil-borne N_2O and CH_4 emissions. The DNDC model is based on equations from classical laws of physics, chemistry and biology and from empirical laboratory observations (Li *et al.*, 2006). The model was first developed to simulate carbon and nitrogen flows in agricultural soils (Li *et al.*, 1992). Since then, it has been refined and updated by a number of researchers across the world to fit specific research purposes, which has resulted in branching of the model (Gilhespy *et al.*, 2014). Here, a Canadian version (DNDC-CAN) developed and validated for similar cool-weather conditions as those prevailing in Sweden was used. This version has been refined to *e.g.* better reproduce crop biomass growth (Kröbel *et al.*, 2011) soil temperature (Dutta *et al.*, 2017) and evapotranspiration (Dutta *et al.*, 2016). DNDC-CAN has also recently been extended to simulate perennial regrowth after cuts in subsequent years (He *et al.*, 2019). The model has been used in previous LCA studies to simulate impacts of agricultural systems (Goglio *et al.*, 2014, 2018).

In Papers I and II, the DNDC-CAN model was fed with site-specific data comprising soil properties, climate, location and management set-up. Data on soil porosity, density, field capacity and wilting point were obtained using a pedotransfer model developed by Saxton & Rawls (2006). The model fit to observed biomass growth data was analysed in Paper I.

In Paper II, the same model set-up as employed for the crop in Paper I was used, but the grass was assumed to be cultivated on fallow land in Uppsala municipality. Based on measuring points comprising carbon, clay, sand and silt content, as well as pH, the study sites were given specific properties with Geographic Information System (GIS) programming.

4.3 Climate impact assessment

When all crucial data have been collected in the life cycle inventory, the next step in LCA is to estimate the environmental impact caused by these emissions.

In Papers I and II, both GWP and the dynamic climate impact model were used (see section 3.3 of this thesis). All major fluxes of the GHGs CO_2, CH_4 and N_2O during the life cycle (see sections 4.1.1 and 4.1.2) were included. In Paper II, the climate impact of the biogas per MJ was compared against the impact of a fossil alternative, diesel fuel. This was calculated as:

$$GWP\,reduction = (GWP_F - GWP_B)/GWP_F \qquad (3)$$

where GWP_B is the GWP caused by net emissions from the studied system, without fossil fuel substitution (*i.e.* E_{Tot} - E_{FossR}), and GWP_F is the GWP caused by emissions from an equivalent amount of fossil fuel (E_{FossR}).

4.4 Eutrophication assessment

In Paper I, the eutrophication impact of grass cultivation was assessed at the five sites. This was done using nitrogen and phosphorus leaching data from Johnsson *et al.* (2016), who calculated mean leaching rates through simulations for the 22 regions in Sweden. The data are presented for specific crops and soil textures and include leaching from the root zone and surface runoff.

The most common approach used for assessing eutrophication is the CML method (Guinée, 2002). This is a site-generic method, which places the impact indicator at the point of emissions and hence neglects the fate of the eutrophying emissions. Furthermore, the method does not consider whether the recipient is nitrogen- or phosphorus-limited, and therefore all discharges of nitrogen and phosphorus to the environment are considered to be potentially eutrophying. This is a simplistic approach and in reality eutrophication is a much more complex phenomenon. This is especially true in Sweden, which is surrounded by the Baltic Sea, the world's largest brackish water basin. The Baltic Sea is considered both nitrogen- and phosphorus-limited, with the degree varying between different sub-basins (Swedish EPA, 2006). Therefore, to complement the CML method, a site-specific method for assessing marine eutrophication was used in Paper I. This method was developed by Henryson *et al.* (2018), who present emissions factors for different regions in Sweden. The emission factors used in CML and the Henryson approach are presented in Table 2.

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*

Site	Marine eutrophication (Henryson et al.) (kg N-eq kg^{-1})		Potential eutrophication (CML) (kg N-eq kg^{-1})	
	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

4.5 Energy balance assessment

In Paper II, the energy balance of the grass-based biogas system was evaluated by calculating the energy ratio (ER) (Djomo *et al.*, 2011), using the equation:

$$Energy\,ratio = E_{OUT}/E_{IN} \qquad (4)$$

where E_{OUT} is the energy produced in the system, biogas in this case, and E_{IN} is the primary energy input to the system in terms of fossil fuel and electricity. The fraction of the biogas produced that was used to heat the reactor was not included in the energy balance calculations.

5 Results and discussion

5.1 Climate impact

5.1.1 Grass cultivation

In the analysis in Paper I, the climate impact of grass cultivation showed large variability between the five sites and between the two fertiliser intensity levels (Figure 8). According to the results, soil properties and weather conditions were more important than fertiliser intensity for the climate impact of grass cultivation. For all sites, the higher fertilisation rate (200 kg N ha^{-1}) entailed a lower climate impact per Mg harvested biomass. The GHG fluxes with the largest climate impact were changes in the soil carbon stock, soil N_2O emissions and emissions from fertiliser manufacture. The changes in soil carbon stock mitigated the climate impact of grass cultivation, while N_2O emissions from soil and emissions from fertiliser manufacture increased the climate impact. The largest climate impact per Mg DM yield was found for the fine-textured soil in Kungsängen, at the lower fertilisation rate. Lower climate impact was found for the coarser-textured soils in Klevarp and Tönnersa with lower initial carbon content. A lower impact was also found at the Lanna site, which had the largest soil carbon sequestration.

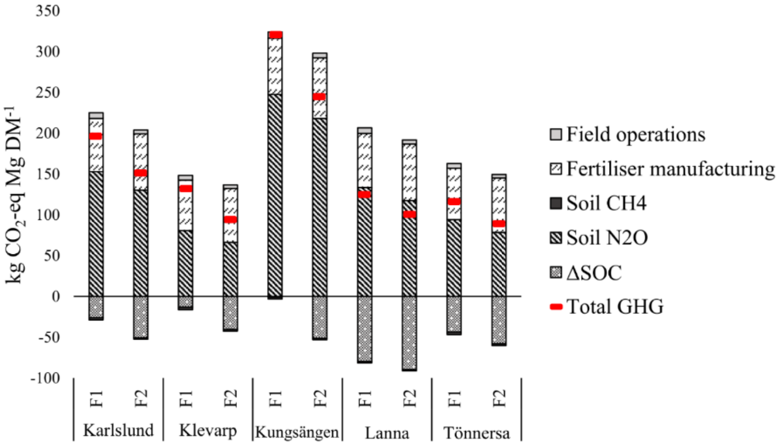

Figure 8. Total climate impact of grass cultivation during 30 years at the five study sites in Paper I for fertilisation rates F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}), assessed as Global Warming Potential over 100 years (GWP_{100}).

5.1.2 Grass-based biogas

In Paper II, the system boundary was expanded to include utilisation of the harvested grass biomass and, in contrast to Paper I, the net effect of the system was investigated. Figure 9 shows the climate impact of the system over the studied time horizon using all fallow land included in the analysis. In the land-use compartment (ΔLU), the largest impact was from nitrogen fertiliser manufacture and soil-borne N_2O emissions, while a negative impact was simulated in terms of soil carbon sequestration, which is in accordance with the results in Paper I. However, when the net effect of grass cultivation was studied the largest impacts were from nitrogen fertiliser manufacture, mainly since no fertiliser was used in reference land use, green fallow. In contrast to Paper I, field operations resulted in a relatively larger climate impact. This was because chopping of the grass biomass was included in Paper II to facilitate subsequent anaerobic digestion. In the fuel production compartment (ΔFP), the largest impact was due to CH_4 losses during production and storage of the digestate. This compartment had a large net negative impact from the avoided emissions from the substituted diesel fuel. For the soil fertilisation compartment (ΔSF), the largest mitigation potential was in increased soil carbon stock and avoidance of nitrogen fertiliser manufacture through substitution of digestate.

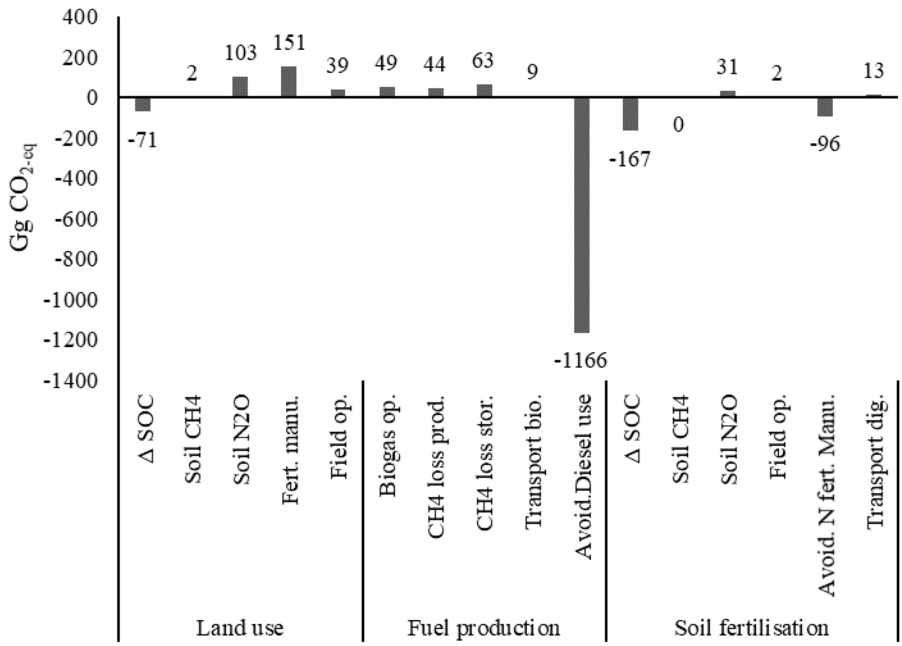

Figure 9. Total net climate impact of the grass-based biogas system in Uppsala municipality over 100 simulated years. The emissions (assessed as GWP_{100}) are aggregated into the three compartments: Land use, Fuel Production and Soil fertilisation, and are expressed in Gg CO_2eq (G = 10^9)

The climate impact was also assessed dynamically over the study period. The net climate impact of the total grass-based biogas system showed a decreased temperature response over the study period (Figure 10). Although the altered system entailed an increased temperature response, the impact from the reference system was larger, which resulted in a total net negative climate impact. This was largely attributable to the substitution of diesel fuel. For the altered system, the impact was dominated by the emissions from $GrassC^A$ in the long-term and $BioC^A$ in the short-term. This was because the main emission from $BioC^A$ was CH_4, which is a relatively short-lived climate forcer. This explains the climate impact declination for this subsystem over time. For the net effect of the system, the land use compartment (ΔLU) and the soil fertilisation compartment (ΔSF) more or less cancelled each other out over the 100-year time horizon.

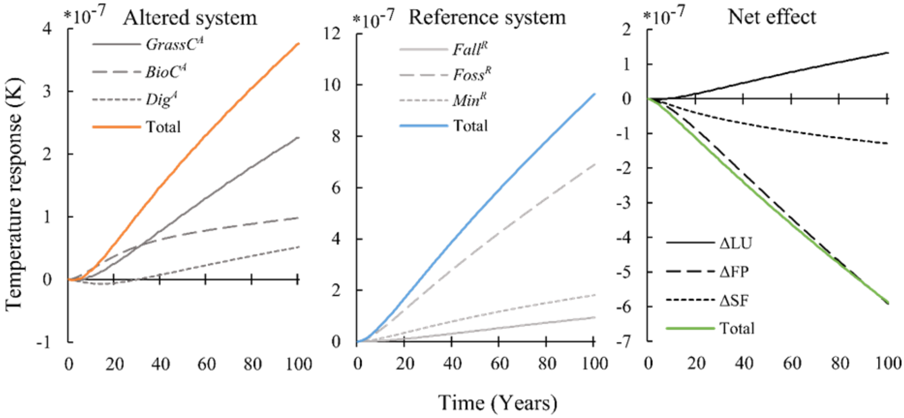

Figure 10. Temperature response, in degrees Kelvin (K) using all fields studied (N=1240, 3006 ha) in the region, for (left) the altered system and (centre) the reference system, and (right) the total net effect.

When all study sites in the region were included, the net GWP of the biogas produced was 10 g CO_2-eq MJ^{-1}, without substitution of fossil fuel (*Foss*^{R}). This corresponded to a GWP reduction of 85% compared with diesel fuel. However, the variation between sites was large, ranging between 102 and 79% reduction, depending on where in the region the grass was cultivated. Figure 11 shows the GWP reduction compared with diesel fuel depending on the fraction of total available land area used. For example, if only 10% of the best performing land was utilised, the GWP reduction increased to 95%.

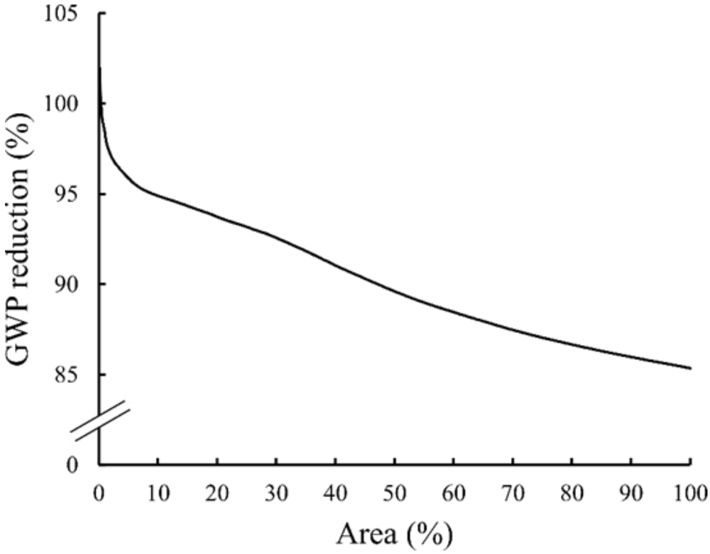

Figure 11. Global warming potential (GWP) reduction compared with diesel from using the grass-based biogas system, without fossil fuel substitution (*FossR*), in relation to fraction of total available land area used.

In Paper II a scenario analysis was performed, where the base scenario was compared with two alternative scenarios: (i) increased fertilisation in grass cultivation and (ii) biomass cultivation with biological nitrogen fixation. Increasing the fertilisation intensity increased the total climate mitigation effect of the system compared with the base scenario. This was attributable to the increased yield in scenario (i), which entailed higher biogas production, enabling more diesel to be substituted. The biological nitrogen fixation scenario (ii) also resulted in total higher mitigation potential than the base scenario, although the yield was lower. This was because no fertiliser was used, due to the assumption that the nitrogen requirement of the crop was met through atmospheric nitrogen fixation. This scenario also reduced soil N_2O emissions, which corresponds with IPCC default values for leguminous crops (IPCC, 2006), where direct N_2O emissions are neglected based on results from Rochette & Janzen (2005). The climate impact per MJ produced biogas of the two scenarios was also analysed. The results indicated that the nitrogen fixation scenario yielded the most climate-efficient biofuel, with higher mitigation effect than both the base scenario and the increased fertilisation scenario. The lowest mitigation effect per MJ biogas produced was in the increased fertilisation scenario (Figure 12).

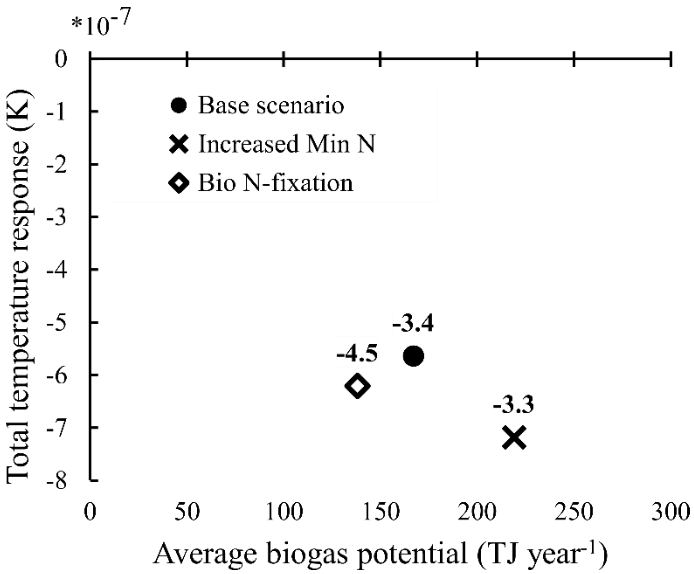

Figure 12. Total temperature response (degrees K), over 100 years, and average biogas potential (TJ per year) for the base scenario and for two alternative scenarios: increased mineral nitrogen (Min N) fertilisation and use of biological N-fixing crops. The values next to the points show the temperature response per unit of biogas produced ($K*10^{-17}MJ^{-1}$).

5.2 Soil carbon balance

5.2.1 Grass cultivation

Over the 30-year time horizon applied in Paper I, the carbon stock increased for all sites and with both fertiliser intensities, though the change was small for Kungsängen F1 (Figure 13). Increasing the fertilisation intensity from 140 to 200 kg N ha^{-1} increased carbon sequestration at all sites, although there were fluctuations in the curves as a result of the rotation period (Figure 13). In every fifth year the grass was restarted, *i.e.* the standing grass was ploughed under, which meant that all biomass, above and below ground, entered the soil organic carbon pool. For most sites, the sequestration rate was higher in the first part of the simulation period. This is worth considering when including soil carbon in climate impact assessments, since the period over which soil carbon change is averaged may have a large impact on the calculated sequestration. The total sequestration over the 30-year time horizon varied between 0 and 4 for the F1 fertilisation intensity (140 kg N ha^{-1}) and between 3 and 6.5 Mg C ha^{-1} for the F2 intensity (200 kg N ha^{-1}). This shows that not only the fertilisation rate, but also the spatial properties of the site, are important in determining soil carbon sequestration. According to Bolinder *et al.* (2017), the mean soil carbon sequestration potential of grass cultivation in Sweden is 560 and 85 kg C ha^{-1} y^{-1} in the topsoil and subsoil, respectively. However, these values represent the net effect compared with cultivating annual cereals, whereas in the results presented in Figure 13 only the gross effect is shown. Depending on what is chosen as the reference scenario, the net effect will display large variation.

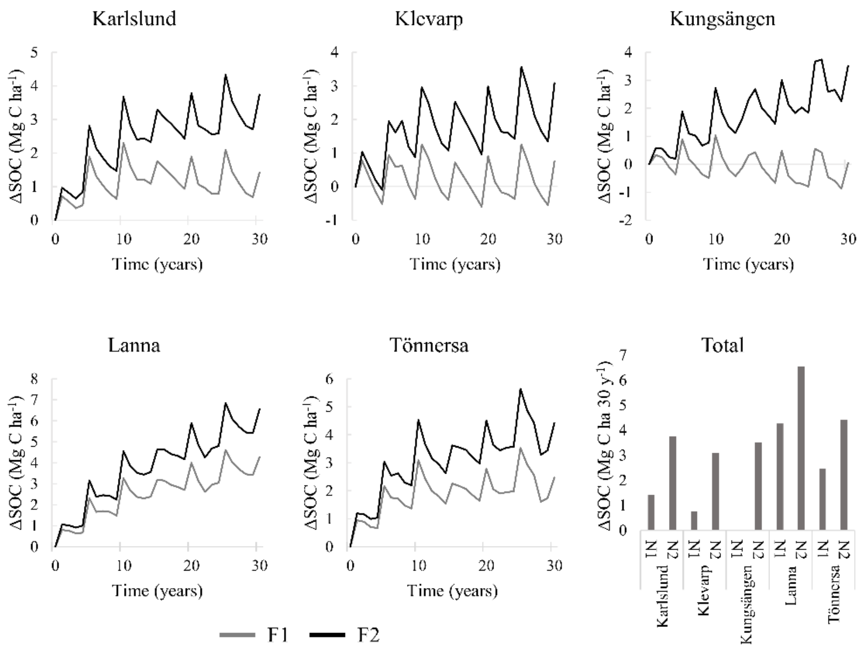

Figure 13. Simulated cumulative change in carbon (C) stock for the five sites investigated in Paper I and the total for all sites. The grey line represents fertilisation rate F1 (140 kg N ha^{-1}) and the black line F2 (200 kg N ha^{-1}).

5.2.2 Grass-based biogas

In Paper II, the grass was assumed to be cultivated on fallow land in Uppsala municipality. The net soil carbon balance for the grass cultivation was calculated as the difference between grass cultivation and fallow land (Figure 14).

The carbon balance for grass cultivation showed large spatial variability in the region, where the introduction of grass cultivation led to increased soil carbon stock at some sites and carbon depletion at others. However, most soils showed an increase in carbon concentration with grass cultivation. A similar pattern was found for the reference fallow land, but the carbon increase was smaller and the depletion was larger. This led to a net increase in carbon stock at all sites, which means that 100 years of grass cultivation would result in higher carbon concentration at the sites than 100 years with fallow. The net effect of the soil carbon change showed lower spatial variability, due to counterbalanced spatial variability in grass cultivation and fallow land. The importance of the dynamic dimension of the soil carbon balance was more evident in Paper II, where a 100-year perspective was adopted, than in Paper I with its 30-year study period.

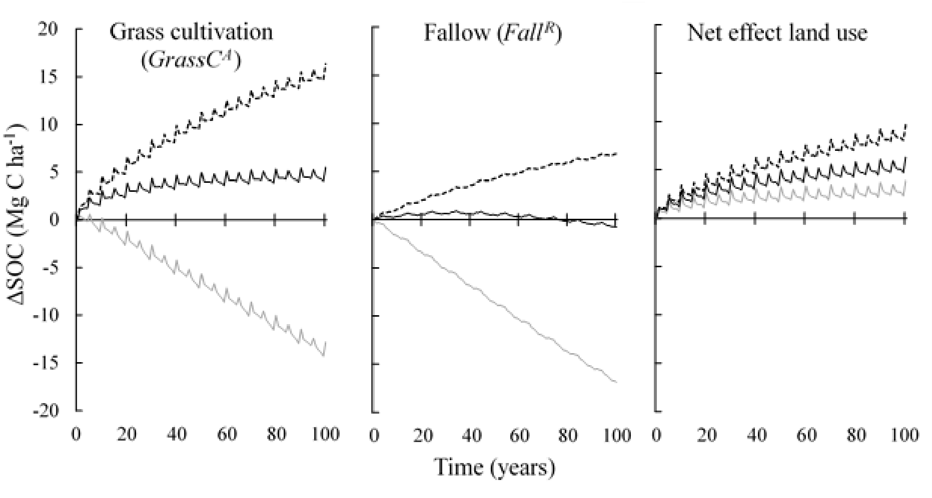

Figure 14. Simulated cumulative change in soil carbon (C) sequestration in the study region of Uppsala municipality. Change in C stock (Mg C ha^{-1}) for (left) grass cultivation only and (centre) fallow only, and (right) net effect of the land use, *i.e*. grass cultivation – fallow. The grey line represents the 5th percentile in the region, the black line the median and the dashed black line the 95th percentile.

The spatial variability in carbon stock change in grass cultivation showed the highest correlation to initial carbon content (r = -0.79) and clay content (r = 0.50). This indicates that soils with low initial carbon content and high clay content had a higher capacity to sequester carbon (Figure 15). Similar results have been obtained in other studies (Kätterer *et al.*, 2012; Poeplau *et al.*, 2015b).

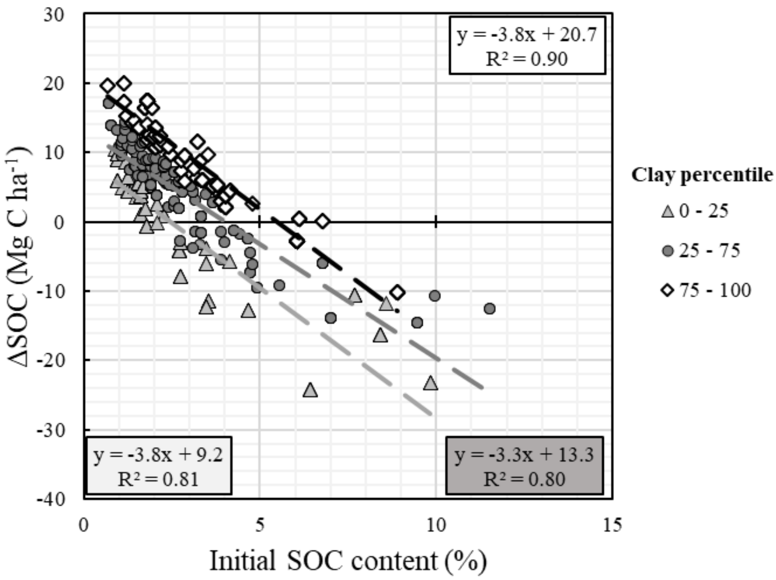

Figure 15. Change in soil carbon (C) content (%) after 100 years of grass cultivation, related to the input parameters with the largest correlation for the study region, initial C content and clay content. Total C change from grass cultivation on the y-axis, initial C content on the x-axis. The colour and shape of the markers represent the clay content. Fitting line and R^2 for the respective clay concentration is displayed in the corners, 0-25th percentile in the lower left corner, 25-75th in the lower right corner and 75-100th in the upper right corner.

In Paper II, the effect on soil carbon content of using the digestate from the biogas production as fertiliser in winter wheat cropping was analysed. The results showed increased soil carbon stock with digestate application and large depletion of the carbon stock with mineral fertilisation. This resulted in large net soil carbon sequestration for the soil fertilisation part of the life cycle. Compared with the ΔLU compartment, the net effect of using digestate as fertiliser showed a greater net increase in the soil carbon stock in the ΔSF compartment (Figure 16).

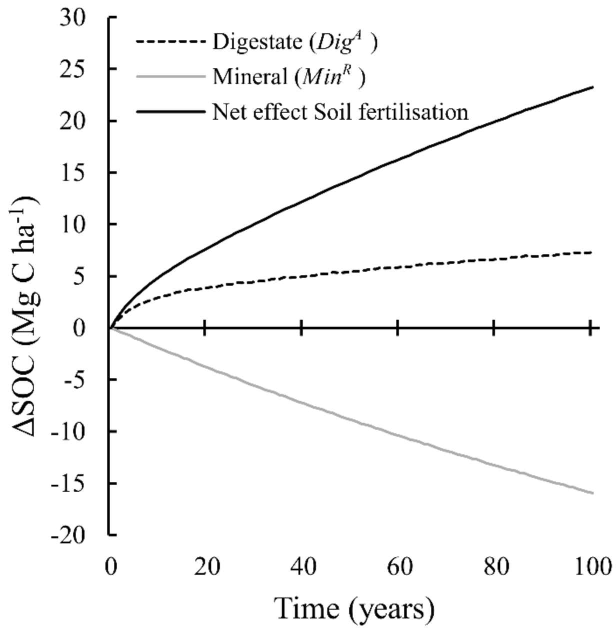

Figure 16. Simulated soil carbon (C) sequestration in winter wheat cultivation using mineral fertiliser (grey), digestate fertiliser (dashed) and the net effect, *i.e.* digestate – mineral. Modelled for one field, which represented average conditions in the study region

This was mainly because of the high carbon depletion in winter wheat cultivation with mineral fertiliser. Unfortunately, studies regarding the long-term impact of digestate on soil carbon are lacking. Digestate is a complex organic amendment, since its composition varies between biogas systems depending on the substrate used. Digestate could potentially be compared to sewage sludge or manure (Bolinder *et al.*, 2017). Results from long-term field trials on degradation of farmyard manure showed that the carbon fraction remaining in the soil after 5, 10 and 37 years was 30%, 20% and 9%, respectively (Tatzber *et al.*, 2012). Using these figures for farmyard manure degradation, the soil carbon sequestration from digestate application would be 10 Mg ha^{-1} over 37 years, which indicates that the estimates obtained in this thesis may be slightly low.

5.3 Soil nitrous oxide emissions

5.3.1 Grass cultivation

For the LCA calculations, the soil-borne N_2O emissions were estimated using the DNDC model. In Paper I, the yearly cumulative emissions showed large variation between sites and fertiliser intensities. Increased fertilisation elevated the mean N_2O emissions for all five sites. The highest emissions with both F1 and F2 were from the soil in Kungsängen, while the lowest emissions were from the Klevarp site (Table 3). In general, the N_2O emissions were higher from fine-textured soils than from coarser-textured soils. This is consistent with findings from a meta-analysis based on observations by Rochette *et al.* (2018).

Table 3. *Simulated yearly mean N_2O emissions for the five sites studied in Paper I, estimated using the DNDC model (mean over a 30-year time horizon ± standard deviation)*

Site and fertilisation intensity	Mean N_2O emissions (kg N_2O ha^{-1})	
	F1	F2
Karlslund	3.4 ± 0.46	3.9 ± 0.54
Klevarp	1.9 ± 0.27	2.1 ± 0.32
Kungsängen	5.2 ± 0.95	6.1 ± 1.03
Lanna F1	2.9 ± 0.34	3.5 ± 0.40
Tönnersa F1	2.2 ± 0.35	2.5 ± 0.42

The N_2O emissions simulated with the DNDC model were compared against emissions estimates obtained using two other methods, IPCC tier I and the method developed by Rochette *et al.* (2018). The results showed only small differences between the mean values obtained with the different methods. However, the estimates obtained with the site-specific methods DNDC and Rochette *et al.* showed large variation in emissions between sites and years. As shown in Table 3, the DNDC model estimated the highest emissions for the soil in Kungsängen, whereas the Rochette *et al*. method estimated the highest emissions for the soil in Lanna. The estimates for the remaining sites were quite similar, but with generally higher estimated emissions with the DNDC model.

5.3.2 Grass-based biogas

In Paper II, the N_2O emissions were simulated in the same manner as in Paper I, but the simulation was performed for over 1000 sites in Uppsala municipality and compared with a reference system, *i.e*. fallow. For the grass cultivation, the mean yearly N_2O emissions varied between 4.4 and 0.6 kg N_2O ha^{-1}, with 3.2 kg

N_2O ha^{-1} from the median soil (Figure 17). The high proportion of fine-textured soils and high soil carbon content in the region could explain the rather high N_2O emissions. This led to high N_2O emissions also from the fallow land, ranging between 3.7 and 0.3 kg N_2O ha^{-1}, with 1.3 kg N_2O ha^{-1} from the median field. This resulted in all sites having net N_2O emissions varying between 2.0 and 0.2 kg ha^{-1}, depending on where in the region the grass was cultivated. The increase in N_2O emissions compared with the fallow land was attributable to the mineral nitrogen fertiliser applied in grass cultivation. The dynamic variations in N_2O emissions were due to variations in the input weather data. In Paper II, 10-year weather data were looped in the model, which explains the recurring pattern in the emissions (Figure 17). The modelled spatial variation in N_2O emissions from grass cultivation showed the strongest correlation to soil pH ($r = -0.87$) and initial soil carbon concentration ($r = 0.50$), indicating that soils with low pH and high carbon content were most likely to cause high 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. This effect has been attributed to the interference from N_2O denitrification in environments with lower pH (*e.g.* McMillan *et al.*, 2016; Russenes *et al.*, 2016).

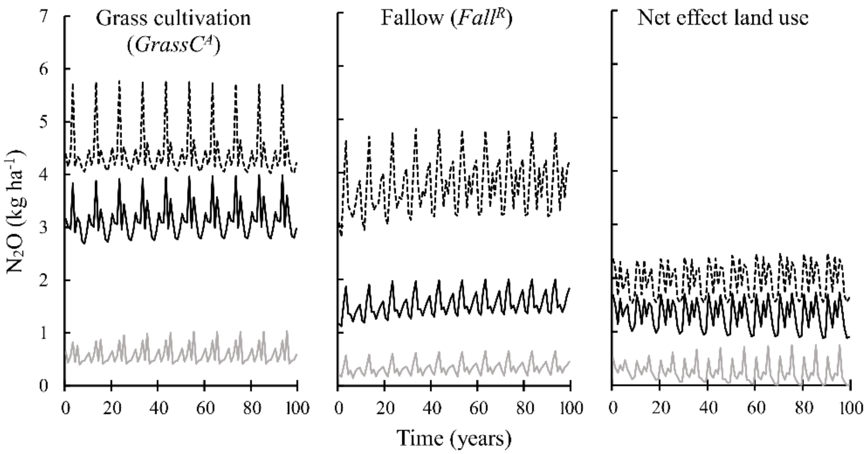

Figure 17. Annual nitrous oxide (N_2O) emissions from soil in (left) the grass system and (centre) from fallow land, and (right) net emissions from biogas feedstock cultivation during 100 years for all sites (N=1240). The dashed black line represents the 95th percentile soil (max), the grey line the 5th percentile soil (min) and the black line the median soil.

The net effect on total N_2O emissions of using digestate instead of fertiliser was also investigated. Mean net N_2O emissions were 0.5 kg N_2O ha^{-1} y^{-1}, *i.e.* use of digestate in winter wheat cultivation entailed on average higher N_2O

emissions than with mineral fertiliser. However, the variation between years was quite large throughout the 100-year study period, with somewhat increasing emissions from digestate use and decreasing emissions from mineral fertiliser use over time. The increased N_2O emissions from the digestate simulations were attributable to the increased nitrogen content in the soil due to increased availability of degradable soil organic matter from the digestate (Figure 18).

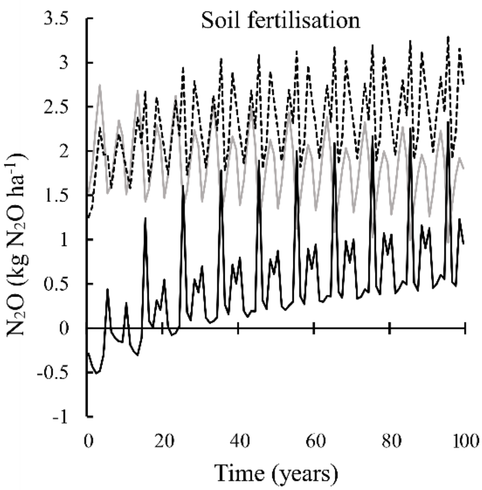

Figure 18. Annual nitrous oxide (N_2O) emissions from soil over a 100-year time horizon following fertilisation of winter wheat. The grey line represents emissions from winter wheat cultivation with mineral fertiliser, the black dashed line emissions from winter wheat cultivation with digestate. The black line shows the net effect, *i.e*. the difference between digestate and mineral fertiliser.

5.4 Eutrophication

The eutrophication impact of grass cultivation was assessed in Paper I, using both the CML method, which assesses potential eutrophication, and the method developed by Henryson *et al.* (2018), which assesses marine eutrophication (Figure 19). The largest potential eutrophication effect was found for the coarser-textured soils in Klevarp and Tönnersa, due to higher leaching rates, while the fine-textured soils had lower impacts. For assessment of marine eutrophication, the location of the site was included. The results indicated the largest eutrophication for grass cultivation in Tönnersa, which was due to high nitrogen leaching rate and proximity to the coast (see Figure 3). These two methods for assessing eutrophication should not be compared to each other, since they describe different types of eutrophication. Instead, they can be seen as complementary approaches.

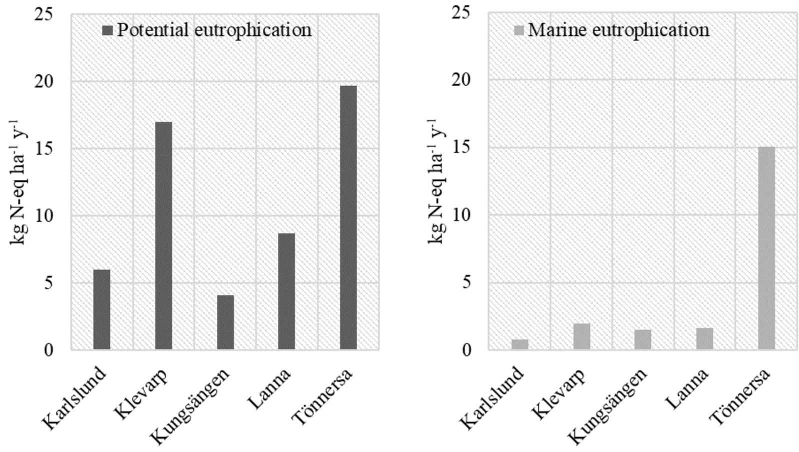

Figure 19. (Left) Potential eutrophication impact assessed using the CML approach and (right) marine eutrophication impact assessed using the methodology of Henryson *et al*. (2018)

5.5 Energy balance

In Paper II, the energy balance of the altered system was analysed by applying the energy ratio equation, where the energy output in terms of biogas was divided by the primary energy input in terms of fossil fuels and electricity. The largest primary input was found to be in the biomass conversion subsystem, where most energy was used for upgrading, compression and 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 47.8 TJ y^{-1} and the energy output was 167.4 TJ y^{-1}, which resulted in an energy ratio of 3.5 (Figure 20).

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

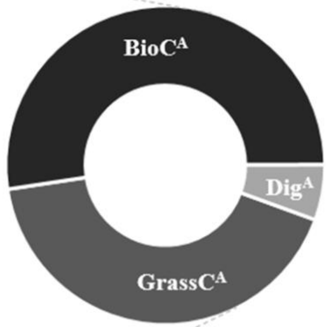

Figure 20. Annual primary energy (PE) input and energy output of the altered system for the study region of Uppsala municipality, divided between the subsystems grass cultivation (*GrassC^A*), biomass conversion (*BioC^A*) and digestate use in winter wheat cultivation (*Dig^A*).

5.6 Concluding discussion

5.6.1 Grass-based biogas and climate mitigation

The results from Paper II showed that implementing the proposed system, where fallow land was converted to grass cultivation for biogas production, resulted in a considerable climate impact reduction for the study region (Figure 10). As for most energy production systems, the grass-based biogas entailed gross environmental impacts. The climate impact was mostly influenced by soil-borne N_2O emissions and emissions from fertiliser manufacture. The calculated net climate mitigation effect was very dependent on the reference system used. In Paper II, biogas was assumed to replace diesel fuel, which entailed large negative GHG emissions (Figure 12). In the current Swedish context, using biogas to replace diesel for transportation purposes is a realistic option. However, over a

100-year perspective the most obvious reference fuel may change, which was not considered in the assessment in Paper II.

The most commonly used method to assess climate impact in LCA is to calculate GWP. One problem with this method is that it does not include the dynamic variation in the impact. Thus the dynamics of the impact were also analysed in this thesis. The results revealed that the impact of different processes changed over time. This aspect of climate impact can be important when planning agricultural systems. For the grass-based biogas system studied, the biogenic CO_2 from combusting the biofuel was not included because the grass had a short rotation. For a feedstock with a longer time between harvests, biogenic CO_2 can also be included in the analysis to show the payback time of the biofuel.

In general, grass cultivation increased soil carbon stock. The grass-based biogas system yielded double net soil carbon sequestration, both from the grass cultivation itself and from using the digestate in winter wheat cultivation (see *Figure 9*). These results confirm the importance of including soil carbon balance in climate impact calculations of biofuels. Under current EU regulations, biofuels that meet the sustainability criteria set in the EU Renewable Energy Directive are entitled to a vital tax reduction. However, current regulations do not allow soil carbon effects from feedstock cultivation or from use of digestate to be accounted for (Council Directive 2009/28/EC). This decision penalises perennial feedstock crops in favour of annual crops such as maize or cereals.

The results in this thesis demonstrate the multiple benefits of the proposed grass-based biogas system. Besides providing an alternative to diesel, the system also entails more resilient land use, which could maintain or increase soil fertility through increased soil carbon stock for future biomass cultivation. The grass biomass produced could also serve as back-up animal feed in the event of crop failure, *e.g.* due to droughts or heatwaves, since these types of weather events are projected to become more frequent with increased global temperature (IPCC, 2014).

5.6.2 Temporal- and spatial-dependent LCA

The assessed systems showed large variations between sites and years. These types of variations are rarely included in LCA, but the results in this thesis indicate that they need to be tackled to get the full picture of the environmental impact of agricultural systems. Soil N_2O emissions and soil carbon balance showed the largest influence on the spatial variation in the climate impact of the system. In Paper II, gross soil N_2O emissions showed the strongest correlations to soil pH and soil initial carbon content, while soil carbon change correlated most strongly to soil initial carbon content and soil clay content. For the net effect, almost all spatial variation was explained by variations in N_2O emissions, because the variation in soil carbon changes in grass cultivation was cancelled out by the variation in fallow land.

The simulations showed that the carbon sequestration rate was higher during approximately the first one-third of the simulation period and declined over time (*Figure 13* and *Figure 14*). This demonstrates the importance of analysing the temporal aspect when including soil carbon changes in climate impact footprint calculations. Moreover, it is important to remember that soil carbon sequestration is a reversible process, which means that if the land use scheme is interrupted, *e.g*. if grass cultivation is replaced by annual cereal crops, there is a large risk that the sequestered carbon will be lost again to the atmosphere. Therefore, long-term commitment by landowners is needed to create robust mitigation schemes involving soil carbon sequestration, in order to sustain the mitigating effect.

Furthermore, increased grass fertilisation rate resulted in an increase in soil carbon sequestration. However, this may be a precarious strategy for increasing soil carbon stock, since the increased fertilisation rate in this thesis also increased the soil N_2O emissions. When the soil reaches the new carbon equilibrium it will no longer sequester carbon, but the elevated N_2O emissions will continue, which means that the system can transform from climate-mitigating to climate-forcing over time.

5.6.3 Agro-ecosystem modelling and LCA

Temporal- and spatial-dependent LCA entails large data requirements. This was addressed in this thesis by adopting a biogeochemical agro-ecosystem model (DNDC) to fill data gaps in the life cycle inventory. The DNDC model has been applied in studies all over the world, including in LCA studies, with satisfactory results (Gilhespy *et al.*, 2014). Following an assessment of different methods for estimating soil-borne N_2O and CO_2 emissions, Goglio *et al.* (2018) concluded that DNDC was the only model among those tested that gave similar results to measurements for N_2O emissions estimates. An advantage of using these kinds of agro-ecosystem models, in contrast to simple carbon models, is that crop growth and nitrogen and carbon fluxes can be modelled simultaneously, which means that interactions between these processes are included. In this thesis, the DNDC model managed to reproduce observed biomass growth, with positive model efficiency values. This indicates that soil carbon inputs, which affect soil organic carbon turnover, were adequately simulated.

The current paradigm of soil carbon stock change is based on the idea that organic matter stabilisation is controlled solely by the molecular structure of the material. However, this view has been challenged, and consideration of physical protection of carbon in soil has been emphasised (Schmidt *et al.*, 2011). This means that other mechanisms may also be important for the decomposition rate, *e.g*. micro-environmental conditions that restrict the access (or activity) of decomposer enzymes, such as hydrophobicity, soil acidity or sorption to surfaces (Schmidt *et al.*, 2011). Models based on carbon quality are still the best available approach, however, since the understanding of physical protection of soil carbon is in its infancy.

Modelling soil N_2O emissions is associated with large uncertainties. In LCA, the IPCC tier I approach for estimating N_2O emissions is the most common method used. However, IPCC only recommends tier I if other data are lacking (IPCC, 2006), because this approach does not differentiate between spatial properties, *e.g*. regarding soil and weather properties. The modelling approach used in this thesis revealed large spatial variation, which indicates that this aspect has large effects on the life cycle climate impact. In a recent report, IPCC describe a refinement to the suggested approach of greenhouse gas inventories whereby the spatial dimension is included in the IPCC tier I approach by spatially disaggregating emission factors by climate region (IPCC, 2019). This is a step forward in addressing the spatial variation in N_2O emissions, but still on very low resolution.

6 Conclusions

- A grass-based biogas system doubled biogas production in the study region, significantly reduced the climate impact and increased soil quality by increasing soil carbon stock.
- Climate impact assessment of grass cultivation showed substantial spatial variation. The greenhouse gas fluxes with the greatest climate impact were soil N_2O emissions and emissions from fertiliser manufacture and changes in soil carbon balance. Grass cultivation tended to increase soil carbon stock, which reduced the life cycle climate impact of the system, but this effect was highly site- and time-dependent.
- Increasing mineral fertilisation rate increased biogas yield and the climate mitigation potential for the study region, but reduced the mitigation per MJ biogas. Soil properties and weather conditions proved more important than fertilisation rate for the simulated climate impact of grass cultivation.
- The combined LCA-DNDC modelling method could be used to design biomass production schemes in other regions, as a strategic tool to assist in land use planning of local energy production on the most suitable arable land for this purpose.

7 Future research

The method in this thesis could be used to assess the environmental impact from other types of biomass-producing systems for food, feed and fuel production. It could also be used to assess a biomass production system with both food and feedstock crops, *e.g.* a grass-cereal-rotation.

A number of process-based agro-ecosystem models are available, including the DNDC model, all of which are formulated slightly differently. One strategy to deal with the inherent uncertainty in simulations could be to apply different models to the same case and use differences in the results in uncertainty calculations for the LCA outcomes. However, existing models are based on current collective scientific understanding of agro-ecosystem processes and there are still many knowledge gaps that need 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 carbon balance. Field trials with continuous measurements are also needed to better understand the spatial and temporal dynamics of these processes. Field trials are needed in particular to determine the soil carbon balance following digestate application, which could have a large impact on footprinting calculations for biogas production.

The results in this thesis indicate that nitrogen-fixing crops, such as clovers, can be a promising feedstock in bioenergy systems, since they require little or no nitrogen fertiliser, which reduces the life-cycle climate impact of the system. They also result in a lower rate of N_2O formation in soil, which further reduces the climate impact. The nitrogen-fixing crop was simulated rather roughly in this thesis and different aspects of atmospheric nitrogen fixation by crops were not handled in detail. For more comprehensive conclusions, the nitrogen-fixing ability in bioenergy systems needs to be further scrutinised.

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Popular science summary

One strategy to limit global warming is to phase out fossil products and replace them with bio-based alternatives. This is often referred to as transitioning from a fossil economy to a bioeconomy. In Sweden, a major challenge to this transition is the transport sector, where around 80% of fuel used is of fossil origin. Combustion of biofuels is often considered climate-neutral, but all biofuels cause greenhouse gas emissions during production, *e.g.* due to different inputs and land uses. It is important to assess these emissions, since studies have shown that the bio-based alternative can have a greater climate impact than the fossil fuel it replaces. Life Cycle Assessment (LCA) is a comprehensive methodology that aims to include environmental impacts from the whole life cycle of a studied product or system.

Environmental impact of agricultural systems is generated by inputs to the system, such as fertiliser, agro-chemicals, machinery and energy. The efficiency of the system, *i.e.* how much output it produces in relation to input, will also affect its environmental impact. Moreover, in agricultural systems, emissions also occur from biological processes, such as changes in the soil carbon balance and soil N_2O and CH_4 emissions. To complicate matters even further, these emissions are determined by the regional and local climate and by site-specific physical, social and environmental conditions, all of which vary over time and space. Variations in these processes are rarely included in LCA methodology, due to lack of data.

In this thesis, agricultural models were used to include spatial and temporal variability in LCA. In one modelling study, the climate impact and eutrophication impact of grass cultivation at five sites in southern and central Sweden were assessed. The grass was modelled in five-year rotation periods for two different fertilisation intensities over 30 years. In a second modelling study, climate impact and energy balance were assessed for grass-based biogas production over a 100-year period. The grass was assumed to be cultivated on fallow land at more than 1000 different sites in Uppsala municipality, Sweden, harvested twice a year and transported to a central biogas plant for conversion to biogas for transportation purposes.

The results showed that the environmental impact varied widely between sites. Spatial changes in soil and weather conditions had a greater influence on the results than mineral nitrogen fertilisation rate. The greenhouse gas fluxes causing the greatest climate impact were soil N_2O emissions and emissions from fertiliser manufacturing, both of which increased the climate impact of the system, and changes in soil carbon balance, which reduced the impact. The soil carbon sequestration effect of the system showed large spatial and temporal variations, but soils with low initial carbon content and high clay content were generally more likely to show increased carbon stock.

The grass-based biogas system analysed significantly reduced the climate impact from the study region compared with reference scenario (fallow land-diesel fuel-mineral fertiliser). The methodology enables the best sites from a climate impact perspective to be selected, which could reduce the total impact from the biogas system. Analysis of the dynamic impact of the grass-based biogas system revealed that in a short time perspective the climate impact was dominated by biomass conversion, *i.e*. from harvested grass to biogas, while in a longer time perspective the climate impact was mostly influenced by grass cultivation. This effect was because the main greenhouse gas emitted during biomass conversion was CH_4, which is a relatively short-lived climate forcer. This led to the climate impact from this part of the life cycle levelling out over the 100-year time horizon analysed. The grass-based biogas system had a double soil carbon effect, from grass cultivation and from using the digestate as a soil amendment and fertiliser in winter wheat cultivation.

The method developed in this thesis could also be used to study the environmental impact of agricultural systems in other regions.

Populärvetenskaplig sammanfattning

En strategi för att begränsa den pågående globala uppvärmningen är att fasa ut fossila produkter och ersätta dem med biobaserade alternativ. Detta benämns ofta som övergången från en fossilekonomi till en bioekonomi. I Sverige ligger en av de största utmaningarna för att förverkliga denna övergång i transportsektorn, där ungefär 80 % av det bränsle som används har fossilt ursprung.

Alla biobränslen orsakar utsläpp under produktionen av bränslet och de råvaror som behövs. Dessa utsläpp är viktiga att studera eftersom det har funnits fall där biobränslet har visats ha en större klimatpåverkan än det fossila alternativet. Livscykelanalys (LCA) är en metod som tar hänsyn till utsläpp under hela livscykeln av en produkt eller ett system och är därför en passande metod för att beräkna miljöpåverkan av dessa typer av system.

Miljöpåverkan av ett jordbrukssystem beror på systemets inputs, så som gödsel, kemikalier, maskiner, energi osv. Från systemet uppstår även outputs såsom mat, bränsle och foder. Hur effektivt systemet är, dvs hur mycket output som genereras per input, kommer också att påverka den beräknade miljöpåverkan av systemet. För jordbrukssystem sker även miljöpåverkan inom systemet från biologiska processer, exempelvis förändring av markens kollager samt markbundna utsläpp av N_2O och CH_4, vilka är två potenta växthusgaser. För att ytterligare komplicera saker, så är utsläppen påverkade av och inbäddade i klimatologiska, fysiska, sociala och miljöförhållanden. Dessa faktorer varierar dessutom över tid och rum. Variationerna av dessa faktorer är vanligen försummade i LCA, ofta beroende på bristen av data.

I denna avhandling användes jordbruksmodeller för att inkludera rums- och tidsberoendet av miljöpåverkan i LCA-metodiken. Avhandlingen är baserad på två studier. I den första studien undersöktes klimatpåverkan och övergödning av vallgräsodling på fem olika platser, utspritt över södra och mellersta Sverige. Den undersökta vallodlingen simulerades i 5-åriga rotationsperioder, med två olika gödselgivor, och systemet analyserades i 30 år. I den andra studien undersökes klimatpåverkan och energibalansen av vallbaserad biogasproduktion. Vallodlingen antogs vara placerad på mark som rapporterats ligga i träda i Uppsala kommun. Vallen skördades två gånger per år och transporterades till en central biogasanläggning, där biomassan omvandlades till biogas i syfte att användas som bränsle för transporter. I denna studie simulerades vallgräsodlingen på över 1000 olika marker med unika förhållanden, med avseende på jordtyp och transportavstånd. Systemet analyserades över en 100-årsperiod.

Avhandlingens resultat visade att miljöpåverkan av vallgräsodlingen hade stor variation mellan olika typer av marker. I själva verket visade sig spatialt differentierade egenskaper såsom jord och väder ha större påverkan på resultatet än gödselgivan. De växthusgasflöden som hade störst påverkan på systemets klimatpåverkan var utsläpp av lustgas från marken, utsläpp relaterade till mineralgödseltillverkning samt förändring av jordens kollager. De första två flödena ökade klimatpåverkan, medan den sistnämnda, reducerade systemets klimatpåverkan. Inbindning av kol i jordbruksmarken visade dock stor rumslig och tidsberoende variation. Generellt hade marker med låg initial kolhalt och hög lerhalt större benägenhet att öka kollagret.

Implementering av det vallbaserade biogassystemet visade en tydlig reducerad klimatpåverkan, jämfört med det antagna referenssystemet. Med den använda metoden kunde de marker inom regionen som var bäst lämpade för systemet, ur ett klimatperspektiv, identifieras och på så sätt minska klimatpåverkan av den producerade biogasen. Genom att analysera den dynamiska klimatpåverkan avslöjades att för kortare tidsperspektiv skedde den största klimatpåverkan under konverteringen av biomassa, dvs från skördad vall till biogas. Men på längre sikt dominerades klimatpåverkan av vallgräsodlingen. Detta var en följd av att utsläppen i konverteringen av biomassan var främst i form av CH_4, vilket är en relativt kortlivad växthusgas. Detta ledde till att klimatpåverkan från denna del av livscykeln klingade av över den 100-åriga analyshorisonten. För det vallbaserade biogassystemet identifierades en dubbel kolinbindningseffekt, både från vallgräsodlingen och dessutom genom att använda den producerade rötresten som gödselmedel i höstveteodling.

Den framtagna metoden i denna avhandling kan även användas för att studera miljöpåverkan av jordbrukssystem i andra regioner.

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

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Abstract

In recent years, sequestration of soil carbon in perennial cropping systems to mitigate global warming has attracted growing interest. Here, we used LCA methodology in combination with the agro-ecosystem model DNDC to assess the climate impact and eutrophication of perennial grass cultivation at five different sites in Sweden. The system was evaluated for two fertilisation rates, 140 (F1) and 200 (F2) kg N ha^{-1}. The DNDC model predicted increased SOC content for all sites, but the system had a net climate impact over the 30-year simulation period (178 ± 77 and 136 ± 59 kg CO_2-eq for F1 and F2 respectively), mainly due to soil N2O emissions and emissions during fertiliser manufacturing. The climate impact varied over time, with low or negative emissions during the first crop rotation and a higher impact at the end of the simulation period. 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 N2O emissions, and for the silty clay loam, due to high carbon sequestration rate. Comparison of three different methods for estimating N2O emissions, the DNDC model, a site-specific empirical approach and IPCC Tier 1 (the most common method used in LCAs), revealed similar overall mean values, but the Tier 1 approach showed low variation between individual sites. The highest eutrophication potential (19.7 kg N-eq ha^{-1} and 16.9) was estimated for sandy loam soils, while sites with finer soil texture had lower eutrophication potential (8.7, 6.0, 4.1 kg N-eq ha^{-1} respectively). Assessment of marine eutrophication in Sweden is complex, because of the adjacent Baltic Sea, which is considered both N- and P-limited, in different sub-basins. Therefore, in addition to the assessment of eutrophication potential, a site-specific characterisation approach was used to determine marine eutrophication potential. The results revealed that one site had by far the highest marine eutrophication potential, due to high N leaching rate and close proximity to the coast. The outcomes of this study show that climate and eutrophication impact of grass cultivation varies widely between sites and, that agro-ecosystem models can be useful tools for completing the life cycle inventory when essential data are lacking.

Keywords: Greenhouse gas emissions, carbon sequestration, nitrous oxide, life cycle assessment (LCA), perennial cropping systems, DNDC model

1 Introduction

Perennial grasses are one of the most commonly grown crops in humid and cold regions, primarily as forage in animal husbandry, although alternative uses such as feedstock for bioenergy production have been proposed (Tilman et al., 2006; Poeplau et al., 2015; Auburger et al., 2017; Carlsson et al., 2017). In Sweden, grass leys occupy about 40% of total arable land (Swedish Board of Agriculture, 2018). 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). In recent years, global warming mitigation through soil C sequestration has attracted growing interest (Smith et al., 2016; Minx et al., 2018) and grass cultivation has been suggested to have great potential to increase soil C stocks (Tidåker et al., 2014; Yang et al., 2018).

Pure grass swards are reliant on fertilisers to promote high biomass yield and achieve high soil C sequestration (Yang et al., 2018). However, there is a potential trade-off, since fertiliser use in agriculture is associated with environmental impacts, primarily global warming and eutrophication. These climate impacts are caused by both fertiliser manufacturing and soil application, the latter by inducing increased terrestrial emissions of the potent greenhouse gas (GHG) N2O (Goucher et al., 2017). Agriculture is also a major contributor to the abundance of eutrophying nitrogen (N) and phosphorus (P) compounds in the environment (Steffen et al., 2015).

Estimates of N2O emissions from soils are associated with considerable uncertainty, due to substantial temporal and spatial deviations and because the underlying processes impacting emissions are still not fully known (Butterbach-Bahl et al., 2013). The Intergovernmental Panel on Climate Change (IPCC) has presented methods for estimating direct and indirect emission of N2O from managed soils. When rigorously documented country-specific emissions factors are lacking, the Tier 1 approach is recommended (IPCC, 2006). 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).

Life Cycle Assessment (LCA) is a comprehensive approach to investigating the environmental impacts of products and services. Originally, LCA was developed as a site-independent tool for industrial processes, but over time has come to be applied to other areas, such as agriculture. In contrast to the impacts of most industrial processes, the environmental impacts of agriculture are influenced by spatial and temporal variations, such as climate and soil properties (Miller et al., 2006). It is therefore important to investigate the system at a number of sites, under different conditions and over time, to determine the variability in the environmental impact. Such assessments are rare, however, because they require an extensive amount of data and since measurements are time-consuming and costly, they are not typically included in the LCA procedure, for which most practitioners rely on databases with low temporal and spatial resolution (Rebitzer et al., 2004). In parallel to the development of the LCA methodology, much effort has been devoted to developing agro-ecosystem models for investigating processes in agricultural soils and plant production. This development work has resulted in a range of different models, e.g. Daycent, Daisy and DeNitrification-DeComposition (DNDC) (Li et al., 1992; Parton et al., 1998; Abrahamsen & Hansen, 2000). DNDC 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 in recent studies to fill data gaps in LCAs (Goglio et al., 2014, 2018). Furthermore, following an assessment of different methods for estimating soil-borne N2O and CO_2 emissions, Goglio et al. (2018) concluded that DNDC was the only model among those tested that gave similar results to measurements for N2O emissions estimates.

In this study, we assessed the potential climate impact and eutrophication of grass cultivation at five sites in Sweden with different characteristics. The agro-ecosystem model DNDC 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 N2O emissions from soil sources have a high degree of uncertainty, we opted to compare three methods for calculating these emissions.

2 Material and methods

Site-specific data for each of the five sites were used to model life cycle inventory data, which was then used to evaluate the environmental impact of the grass cultivation system. Established life cycle impact assessment tools were applied in combination with novel approaches.

2.1 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 the Lanna (58.5°N) was a silty clay loam with lower SOC content (2.0%) than the two soils at higher latitudes and 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 (S1 and S2) in the Supplementary Material.

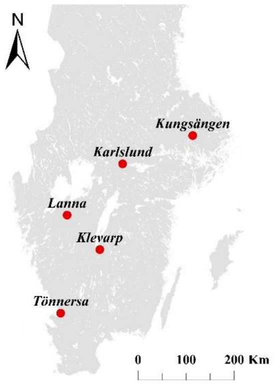

Figure 1. Map of southern and central Sweden indicating the location of the five study sites.

2.2 Perennial cropping system

A five-year grass cultivation system was simulated over 30 years for each of the individual sites. 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.

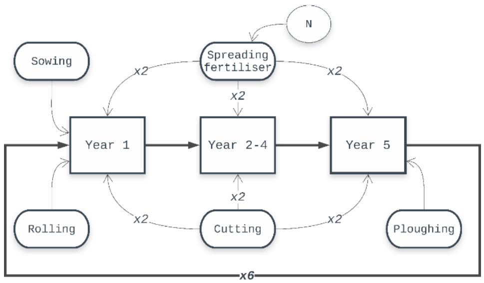

Figure 2. Overview of the crop rotation simulated for all study sites. The grass was sown and the soil was rolled in the first year, then growth continued for five more years. During this period, fertiliser was applied and the grass was cut twice every year. The crop rotation ended with deep ploughing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

2.3 Modelling and assumptions

2.3.1 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 comprises improved 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 improved for simulating perennial regrowth after each cut and in subsequent years (He et al., 2019). This version was chosen because it has been calibrated and validated for perennial growth in similar cool-weather conditions to those in Sweden. 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 (S3). Indirect N2O emissions were calculated using the emission factors from IPCC (2006) for leached and volatilised N, which were simulated with 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 them 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. 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 the Nash-Sutcliffe model efficiency coefficient (Nash & 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.

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

$$nRMSE = (\sum_{t=1}^{n}(O_t - S_t)^2/n)^{1/2}/\bar{O} \quad (2)$$

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

where O denotes observed biomass and S simulated biomass.

2.3.2 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 N2O, 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 N2O-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:

$$N2O_{Roch} = exp(3.91 + 0.0022P + 0.0069MinN - 0.0032SAND - 0.747pH + 0.097T_{air}) \quad (4)$$

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.

2.3.3 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 & 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).

2.3.4 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. They also present data on leaching from the root zone and surface runoff for specific crops and soil textures (Johnsson et al., 2016).

2.4 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., 2013a). 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., 2013a). The characterisation factors for CH_4 and N_2O are 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 that 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).

2.5 Eutrophication impact assessment

The site-generic CML methodology (Guinée, 2002) was used 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). Site-specific marine eutrophication characterisation factors developed by Henryson et al. (2018) for different regions in Sweden, which include site and catchment properties and the P or N limiting status of the recipient, were used 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 (1).

Table 1. 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.

Sites	Marine eutrophication Henryson *et al.* (kg N-eq kg^{-1})		Potential eutrophication CML (kg N-eq kg^{-1})	
	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

3 Results

3.1 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.

3.1.1 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.

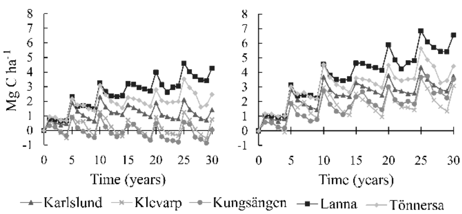

Figure 3. Simulated cumulative change in SOC, for fertiliser rate F1 (a) and F2 (b) over the 30-year period. The SOC change is presented in Mg C ha^{-1}. Fertiliser rate F1 corresponds to 140 kg N ha^{-1} and F2 200 kg N ha^{-1}.

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 with 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 the precipitation. Mean N_2O emissions over the simulation period were slightly higher for the higher fertilisation rate (F2) at all study sites.

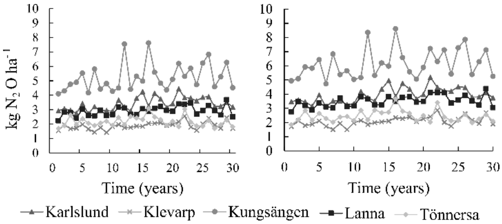

Figure 4. Cumulative yearly N_2O emissions from the study sites (kg N2O ha^{-1}), estimated using the DNDC model, for fertilisation rate (a) F1 and (b) F2 over the 30-year simulation period.

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 5). 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 (S6).

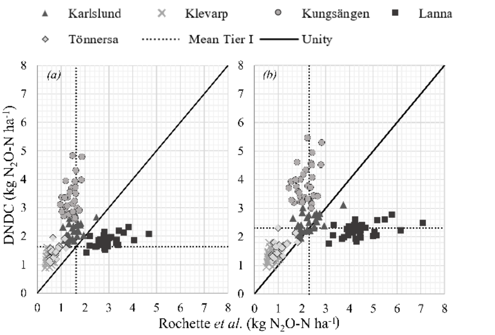

Figure 5. Cumulative yearly N_2O emissions (kg N_2O-N ha^{-1}) estimated with three different methods for fertilisation rate (a) F1 and (b) F2. The results obtained for the Rochette et al. approach are shown on the x-axis and those obtained with the DNDC model on the y-axis. The black line represents unity in the two site-dependent methods and the black dashed line represents mean emissions across all sites calculated with IPCC Tier 1.

3.1.2 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. 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 2).

Table 2. 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

3.2 Life cycle impact assessment

The results from the life cycle inventory were used to assess the climate impact and eutrophication of the grass cultivation system at each of the five study sites.

3.2.1 Climate impact

The GHG fluxes from the inventory analysis were divided into five categories and analysed with GWP_{100} (Figure 6). 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 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.

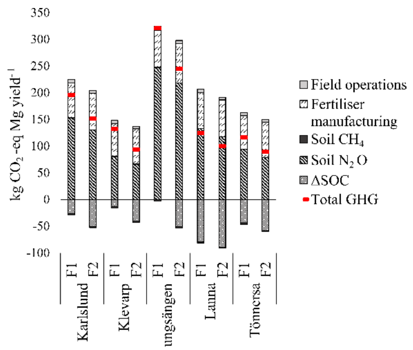

Figure 6. Total climate impact of grass cultivation during 30 years at the five study sites for fertilisation rates F1 and F2, assessed as Global Warming Potential over 100 years (GWP_{100}).

The relationship between the emissions categories shown in Figure (6) 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 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 3). The main reason for this was that the soil C sequestration rate was higher during the first rotation compared with the last.

Table 3. 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}). The results for each site are expressed per Mg DM.

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

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 (7). Similarly to Table (3), 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.

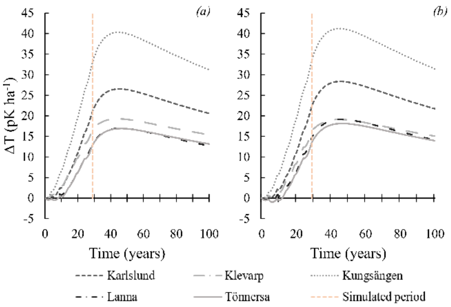

Figure 7. Simulated potential temperature response of the grass cultivation system during 30 years with fertilisation rate (a) F1 and (b) F2 at the five study sites. The temperature response is expressed as pK ha^{-1} (p=10^{-12}, K = Kelvin)

3.2.2 Eutrophication assessment

The potential eutrophication (CML) and marine eutrophication (Henryson et al.) of N and P leaching were assessed on a per-hectare basis (Figure 8). 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.

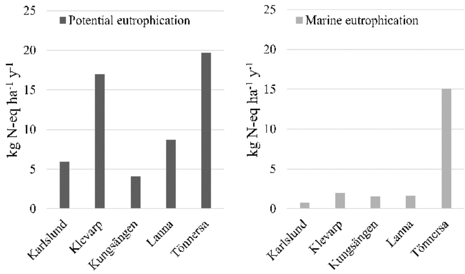

Figure 8. Potential eutrophication impact assessed using CML (potential eutrophication) and Henryson (marine eutrophication) methodology. The bar represents the eutrophication in kg N-eq ha^{-1} y^{-1}.

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 8). The lower impact compared with the CML approach is because the Henryson *et al.* characterisation factors assess marine 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.

3.3 Model biomass growth goodness of fit

The goodness of fit of the DNDC model to the observed data was analysed. Mean simulated aboveground biomass at each cutting occasion was within the standard deviation for each site (Figure 9). 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 S4). Since *ME* was above zero, the model corresponded to the observed data more efficiently than the mean observed value. 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 S5).

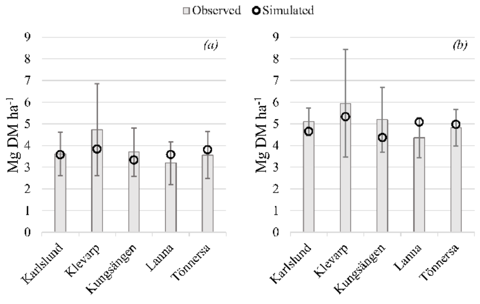

Figure 9. Aboveground mean biomass production at harvest, with fertilisation rate F1 (a) and F2 (b). Bars represent mean of the observations, the rings simulated means and the error bars the standard deviation of the observations.

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

4 Discussion

Assessment of the two impact categories, climate and eutrophication, 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. 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 (Figure 6).

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 e*t 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.

Soil water content and water-filled pore space (WFPS) 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 inserted directly into the model. In contrast, the Rochette *et al.* approach uses soil sand content to describe soil WFPS (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 S1). 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 should be used scarcely in site-specific LCAs.

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. As yearly C sequestration decreased, the climate impact increased, which resulted in increased global mean temperature after both 30 and 100 years (Figure 7). The risk of soil C sequestration schemes transitioning from global warming mitigating to global warming 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 soil-N_2O emissions. 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).

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. 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 neglects 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.

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. This indicates that soil C inputs, which affect SOC turnover, should be adequately simulated.

There are several possible areas of use for the harvested grass biomass. One area is as feedstock in bioenergy production. In Sweden, the annual demand for biofuels is estimated to increase to 20 TWh by 2030. About 8-11 TWh can be supplied using domestic wastes and by-products from forestry, industries, society and animal husbandry. The remaining 9-12 TWh must be supplied by increasing biomass production, e.g. through intensification of grass cultivation, cultivation on fallow land and excess grass silage (Prade *et al*., 2017). Earlier studies have shown that cultivation of perennial woody biomass crops (poplar, willow) on farmland increases the SOC content, which reduces the climate impact of biomass production (Porsö & Hansson, 2014; Hammar *et al*., 2017). The grass cultivation system examined in the present study also showed C sequestering ability. However, to assess the impact of climate mitigating schemes through soil C sequestration, the effect of temporal and spatial variability should be taken into account, as well as non-CO_2 GHG emissions and the eutrophication impact throughout the life cycle. 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.

5 Conclusions

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 the investigated sandy loam and silty clay loam soils.

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.

Mean yearly cumulative N_2O emissions for all study sites were estimated to be 1.97 ± 0.83, 1.41 ± 0.93 and 1.63 ± 0.02 kg N_2O-N ha^{-1} for F1 and 2.29 ± 0.98, 2.13 ± 1.41 and 2.31 ± 0.02 kg N_2O-N ha^{-1} for F2 when estimated with the DNDC, Rochette *et al.* and IPCC Tier 1 approach, respectively. There were only small differences in the results when overall mean values were compared. However, the two site-specific methods, DNDC and Rochette *et al.*, were able to reveal large variations between sites, an effect 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.

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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Supplementary material

Table S1. Specific characteristics of the five study sites. 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.

Site	Karls-lund	Klevarp	Kungsängen	Lanna	Tön-nersa	Source
Latitude	59.4	57.7	59.8	58.5	56.5	(Eckersten *et al.*, 2004)
Mean temp (℃) 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	(Saxton & Rawls, 2006)
Porosity (%)	51	48	48	53	46	(Saxton & Rawls, 2006)
Field capacity (water-filled pore space)	0.67	0.31	0.87	0.72	0.36	(Saxton & Rawls, 2006)
Wilting point (water-filled pore space)	0.38	0.09	0.71	0.39	0.14	(Saxton & Rawls, 2006)

Table S2. Daily precipitation and max-min daily air temperature recorded at weather stations located near each site.

Site	Weather station	Mean annual precipitation (mm) 1986-2015	Mean annual growing season precipitation (mm)	Mean temp (℃) 1986-2015
Karlslund	Örebro airport, Örebro D	691	417	6.8
Klevarp	Jönköping airport, Ramsjöholm	679	411	5.4
Kungsängen	Uppsala airport, Uppsala	568	340	6.9
Lanna	Lanna, Längjum, Hällum A	598	379	7.1
Tönnersa	Halmstad, Genevad, Laholm, Hunnestorp V	791	473	8.0

Table S3. Crop parameters used in the DNDC model. Data in Tables S1 and S2 were used directly in the model. 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 et al. (2016). In the DNDC crop model, the C:N ratio of each biomass component is defined as a fixed value over the growing period. The mean value of C:N ratio for the last observation was used as input to the model.

Parameter	Value	Source
Root:shoot ratio	1	(Johnsson *et al.*, 2016)
C:N ratio aboveground biomass	33.4	Data
C:N ratio belowground biomass	30	(Johnsson *et al.*, 2016)
Thermal degree days for maturity	1900	Adjusted
Water demand (g water g DM^{-1})	200	(Frame & Laidlaw, 2011)
Optimal temperature (℃)	21	(Frame & Laidlaw, 2011)

Table S4. Statistical evaluation of goodness of fit for observed biomass growth at the five study sites using the DNDC model for fertiliser rates 140 kg N ha^{-1} (F1) and 200 kg N ha^{-1} (F2). r^2 = coefficient of determination, RMSE = root mean square error, nRMSE = normalised root mean square error, ME = model efficiency derived with the Nash-Sutcliffe index, n = number of observations.

Sites	F1				
	r^2	RMSE (Mg/ha)	nRMSE (%)	ME	n
Karlslund	0.74	0.91	41	0.54	16
Klevarp	0.52	1.76	56	0.46	34
Kungsängen	0.78	0.71	30	0.77	33
Lanna	0.59	1.08	52	0.34	17
Tönnersa	0.72	1.06	50	0.54	17
All fields	0.61	1.22	49	0.53	117
	F2				
Karlslund	0.71	0.95	30	0.62	17
Klevarp	0.74	1.64	40	0.40	31
Kungsängen	0.64	1.35	40	0.30	33
Lanna	0.77	1.06	36	0.62	17
Tönnersa	0.84	0.86	29	0.73	17
All fields	0.71	1.29	38	0.47	115

Table S5. Statistical evaluation of the goodness of fit for the last biomass observation before harvest using DNDC. r^2 = coefficient of determination, RMSE = root mean square error, nRMSE = normalised root mean square error, ME = the model efficiency derived with the Nash-Sutcliffe index, n = number of observations. Fertilisation rates F1 and F2 correspond to 140 and 200 kg N ha^{-1}, respectively.

Fertilisation rate	Before harvest				
	r^2	RMSE (Mg/ha)	nRMSE	ME	n
F1	0.32	1.35	0.35	0.24	28
F2	0.30	1.50	0.29	0.24	28

Table S6. Mean 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

II

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

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Abstract

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 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 MJ biogas 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 MJ, 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.

Keywords: Grass cultivation, biomethane, soil carbon sequestration, DNDC, regional-LCA, soil N_2O emissions

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. 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 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 (Swedish Board of Agriculture, 2018). 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., 2018, 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, 2018), 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 Material and methods

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, was not included in the assessment. 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) MJ biogas produced. 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 Figure 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 oC. We assumed the same location for the biogas plant as for the current largest existing plant in the region (Figure 1).

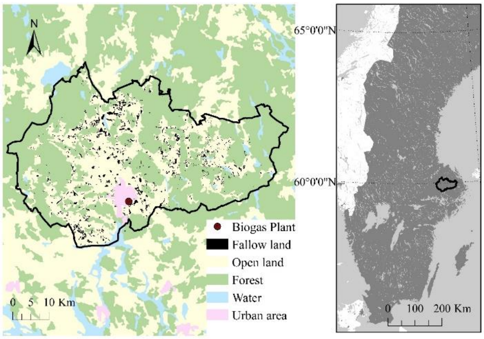

Figure 1. (Left) Map of the study region, Uppsala municipality (inside the black line), showing the distribution of fallow land (black dots) and the location of the biogas plant (red and black purple dot). (Right) Location of the region in east-central Sweden.

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^*) (Figure 2). 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 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 were calculated as the difference between the altered system and the reference system as:

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

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 (Figure 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 calculated to be 2.3, 2.3 and 4.7 dm^3 ha^{-1}, respectively. For cutting, chopping and ploughing, the diesel consumption 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 $Foss^R$ (Figure 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

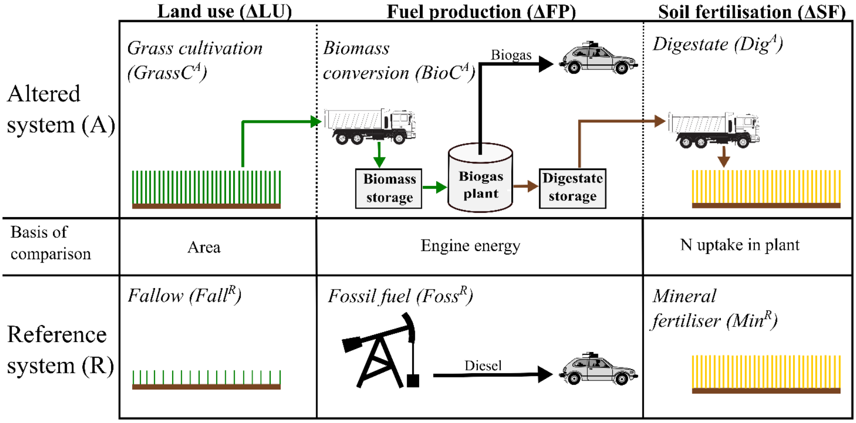

Figure 2. Schematic illustration of the grass-based biogas system studied, divided into six subsystems: Grass cultivation (*GrassC*^{A}), Biomass conversion (*BioC*^{A}), Digestate use (*Dig*^{A}), Fallow (*Fall*^{R}), Fossil fuel (*Foss*^{R}) and Mineral fertiliser (*Min*^{R}). The net effect of the 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} - *Min*^{R}). The basis of comparison is shown in the row between the altered system and the reference system.

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.

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}) (Figure 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 (Figure 3).

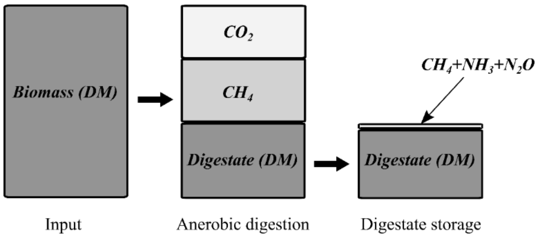

Figure 3. Conceptual model of the mass balance calculation for digestate (illustration not to scale).

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 (Figure 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 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. 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 one hectare of grass cultivation, here denoted Fdig, was calculated as:

$$F_{dig}[\quad] = \left(N_{dig}\left[kg\,\frac{N}{ha}\right] - N_{NH3\,loss\,app}\left[\frac{kg\,N}{ha}\right]\right)/N_{demand}\left[\frac{kg\,N}{ha}\right] \quad (2)$$

F_{dig} was multiplied by the GHG emissions per hectare for the simulated winter wheat cultivation to obtain the GHG balance from the Dig^A subsystem, where N_{dig} is the N in the digestate, $N_{NH3\,loss\,app}$ is the N-NH_3 losses during digestate application, and N_{demand} is the N demand of winter wheat per hectare, 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:

$$Energy\ ratio = Energy\ output/Primary\ energy\ input \quad (3)$$

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. 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:

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

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}).

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., 2016).

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 Figure 4. On average, 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.

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

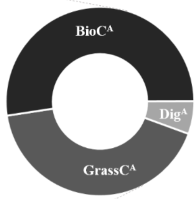

Figure 4. Annual primary energy (PE) input and energy output of the altered system for the study region, divided between the subsystems grass cultivation (*GrassC^A*), biomass conversion (*BioC^A*) and digestate use in winter wheat cultivation (*Dig^A*).

3.1.2 Soil carbon balance

The modelled soil C balance of *GrassC^A* showed large differences between the different sites (Figure 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 (Figure 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.

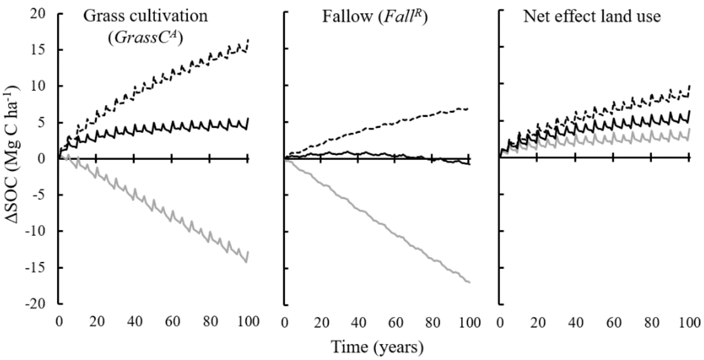

Figure 5. Cumulative change in soil organic carbon (SOC) over 100 years for all sites investigated (N = 1240), simulated with the DNDC model, for (left) grass cultivation only, (centre) fallow land only and (right) the net effect of changing the land use from fallow to intensified grass cultivation.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

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} (Figure 6). On average, the mean digestate produced per ha grass cultivation covered the N demand of 0.66 ha of winter wheat cultivation.

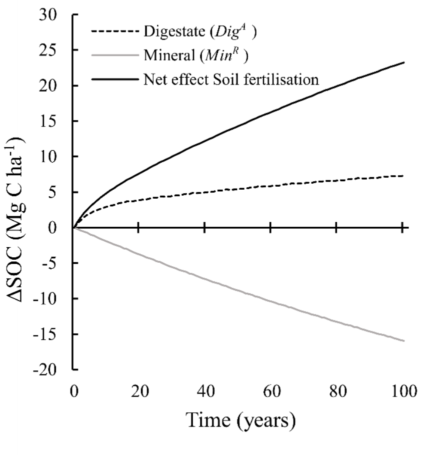

Figure 6. Cumulative change in soil organic carbon (SOC) over 100 years, simulated with DNDC model, for winter wheat cultivation with biogas digestate as fertiliser (dashed), mineral fertiliser (grey) and the net effect, i.e. the difference between digestate and mineral fertiliser (black). The DNDC model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

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 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 *$Fall^R$* (Figure 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).

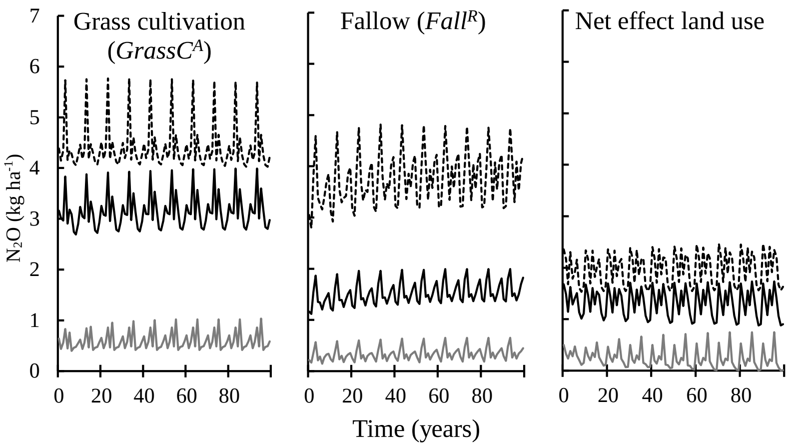

Figure 7. Annual soil nitrous oxide (N_2O) emissions for (left) the grass system and (centre) fallow land, and (right) net emissions for feedstock cultivation during 100 years for all sites investigated (N=1240). The dashed black line represents the 95th percentile soil (max), grey line the 5th percentile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

3.2 Climate impact assessment

The climate impact assessment revealed a net decreased temperature response over the study period (Figure 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 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 (Figure 8).

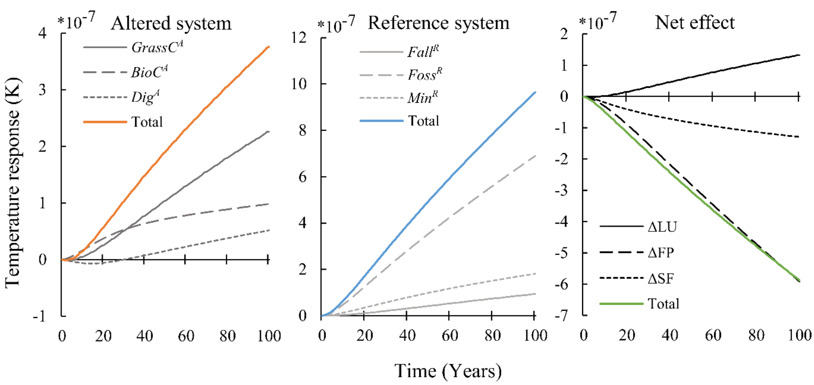

Figure 8. Temperature response, in degrees Kelvin (K) and using all fields studied (N=1240, 3006 ha) in the region, for (left) the altered system and (centre) the reference system, and (right) the total net effect.

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 Figure 9. For instance, if only 10% of the best-performing sites were included, the GWP reduction increased to 95%.

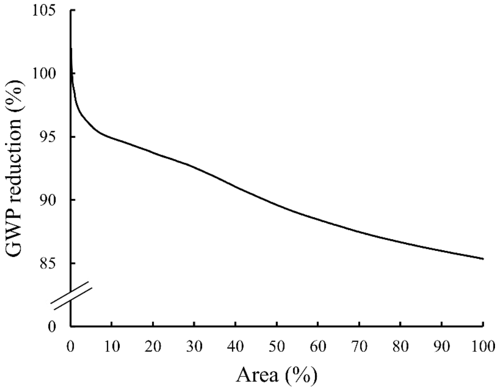

Figure 9. Global warming potential (GWP) reduction compared with diesel from using the grass-based biogas system, without fossil fuel substitution ($Foss^R$), in relation to fraction of total area used in the region.

The spatial difference in the GWP reduction was further investigated (Figure 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 changes soil C stock had a low impact on the spatial variation in climate impact (r = -0.28).

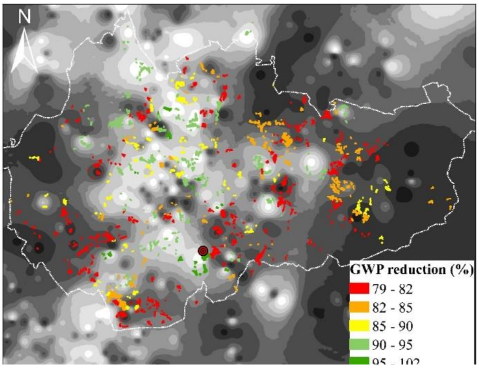

Figure 10. Spatial variation in global warming potential (GWP) reduction compared with diesel of using the grass-based biogas system, without fossil fuel substitution ($Foss^R$). Colours indicate site-specific GWP reduction. Background indicates the soil pH, where a darker shade indicates lower pH. The white dashed line represents the municipality border.

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 Figure 11. The temperature response of the different scenarios was assessed both per MJ biogas produced 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.

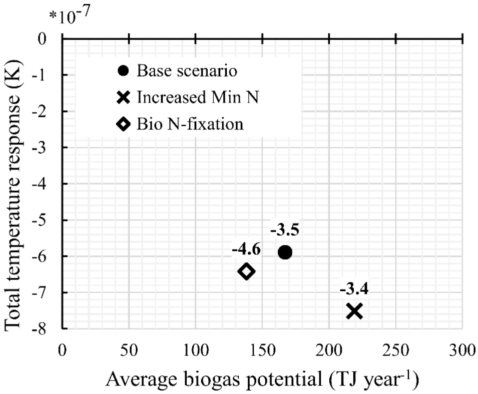

Figure 11. Total temperature response (degrees K), over 100 years, and average biogas potential (TJ per year) for the base scenario and for two alternative scenarios: increased fertilisation and use of biological N-fixing crops. The numbers next to the icons show the temperature response per unit of 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 (Figure 4). Energy ratio for large-scale biogas production in a Swedish context 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 (Figure 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 (Figures 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 (Figures 5 and 6). For instance, the C sequestration rate in grass cultivation with the median soil C change was about fivefold 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 (Figure 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 (Figure 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 (Figure 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 MJ biogas produced. 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 to Uppsala municipality. 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 in Sweden with increased global temperature (IPCC, 2014).

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 with similar conditions, thereby serving as a strategic tool to assist land use planning of local energy production from arable land.

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 MJ biogas. 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.

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Supplementary material

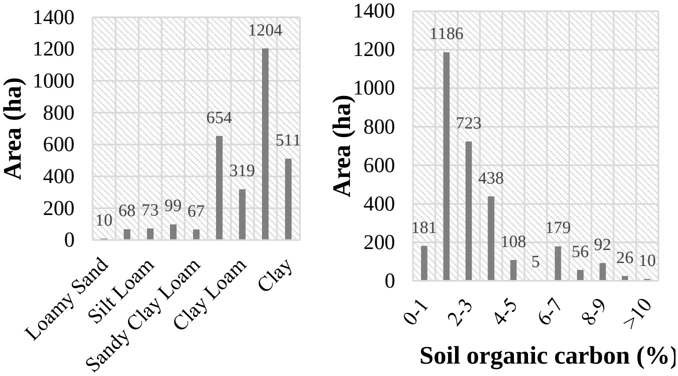

Figure S1. Soil texture and initial SOC content distribution of the reported fallow land in Uppsala municipality.

Table S1. Grass-based digestate properties

	Value
C/N	7.0[a]
Org C (kg ha^{-1})	1183[a]
Tot-N (kg ha^{-1})	169[b]
Org N (kg ha^{-1})	68[a]
NH4-N (% of tot-N)	60[c]
pH	8[d]
Dry matter (%)	9.5[e]

a) Eq S3

b) 135 kg N/MSF

c) (Delin et al., 2012; Sørensen et al., 2011).

d)Assumed value

e) (Carlsson et al., 2017)

Table S2. Crop input parameters to DNDC.

Grass				
	Grain	Leaf	Stem	Root
Biomass fraction	0.01	0.245	0.245	0.5
Biomass C:N ration	34	34	34	30
Thermal degree days for maturity		1900		
Water demand (g water g DM^{-1})		200		
N fixation index (crop N N from soil^{-1})		0		
Optimum temperature degree		21		
Vascularity		0		

Winter wheat				
	Grain	Leaf	Stem	Root
Biomass fraction	0.46	0.185	0.185	0.17
Biomass C:N ration	22	102	102	95
Thermal degree days for maturity		1300		
Water demand (g water g DM^{-1})		200		
N fixation index (crop N N from soil^{-1})		0		
Optimum temperature degree		22		
Vascularity		0		

*Table S3. Pearson correlation analysis of simulated change in SOC content and N_2O emissions (*GrassC^A*) to input site differentiated soil properties initial SOC, clay, sand and silt content as well as pH.*

Pearson correlation coefficient		
Soil properties	ΔSOC	N_2O
Initial SOC	-0.79	0.50
Clay	0.50	0.09
Sand	-0.26	-0.06
Silt	-0.14	-0.01
pH	0.33	-0.87

Calculations

1 Eq S1: Land use (ΔLU)

1.1 GHG emissions from fertiliser manufacturing

```
CO2_from_fertGrass = EF_Nfert_Man * 0.86 * Amount_Nfert;
% CO2 emissions from fertiliser manufacturing, 86% of total CO2-eq
N2O_from_fertGrass = (EF_Nfert_Man * 0.14)/CF_N2O * Amount_Nfert;
% N2O emissions from fertiliser manufacturing, 14% of total CO2-eq

EF_Nfert_Man = 3.6;     % kg CO2-eq/kg N. Emission factor for mineral N fer-
                        tiliser manufacturing, Best Available Technique (Yara
                        International, 2010).
Amount_ Nfert = 140;    % kg N/ha, mineral fertilisation in grass cultivation
CF_N2O = 298;           % CO2-eq (Myhre et al., 2013)
```

1.2 Diesel consumption field operations

```
Diesel_Cutting = (yield*Cut_slope) + Cut_intercept;
% Diesel consumption cutting (L/ha)
Diesel_chopping = (yield*Chop_slope) + Chop_intercept;
% Diesel consumption chopping (L/ha)
Diesel_Ploughing = (clay*Ploughing_slope) + Ploughing_intercept;
% Diesel consumption ploughing (L/ha)
Diesel_spreadFert = 2 * Fac_SpreadFert;
% Diesel consumption spread mineral fertiliser (L/ha)
Diesel_Sowing = Fac_Sowing;
% Diesel consumption sowing (L/ha)
Diesel_Rolling = Fac_rolling;
% Diesel consumption rolling (L/ha)

% Yield = yield (DM) (Mg/ha)
% Clay = soil clay content (%)
Cut_slope = = 0.384 (Prade et al., 2015);
Cut_intercept = 2.334 (Prade et al., 2015);
Chop_slope = 0.268 (Prade et al., 2015);
Chop_intercept = 11.072 (Prade et al., 2015);
Ploughing_slope = 0.32 (Prade et al., 2015);
Ploughing_intercept = 8.5 (Prade et al., 2015);
Fac_SpreadFert = 4.7 (Prade et al., 2015);
Fac_Sowing = 2.3 (Carlsson et al., 2017);
Fac_rolling = 2.3 (Carlsson et al., 2017);
```

2 Eq S2: Fuel production (ΔFP)

2.1 Biomass conversion

```
Prod_upgrad_biogas = yield*VS*SpecCH4*(1 - DM_loss_ens)*Energy_cont_bio;
% Prod_upgrad_biogas = Produced upgraded biogas (MJ)
Energy_use_Heat = yield/DM_cont_feed*Fac_energy_heat;
% Energy_use_heat = produced energy used to heat the plant (MJ)

Output_energy_biogas = Prod_upgrad_biogas - Energy_use_Heat;
% Output_energy_biogas = Net energy produced (MJ)

VS = 0.92;              % volatile solids fraction of DM (Prade et al.,
                        2015);
SpecCH4 = 280;          % Specific methane production grass (m3/Mg VS) (Prade
                        et al., 2008)
```

```
DM_loss_ens = 0.05;   % Yield loss fraction during ensiling (Gissén et al.,
                      2014)
Energy_cont_bio = 34.92;% Calorific value (MJ/Nm3)(Energigas Sverige, 2019)
DM_cont_feed = 0.35   % DM content when fed to the plant (Prade et al.,
                      2015)
Fac_energy_heat = 116.5% (MJ/Mg feedstock)
```

2.2 Operations in biomass conversion

```
Diesel_compaction = yield*Fac_energy_Compaction;
% Diesel consumption compaction (L/ha)
EL_Upgrading = Prod_upgrad_biogas*Fac_energy_Upgrad;
% Electric use upgrading (MJ/ha)
EL_PumpandStir = yield/DM_cont_feed * Fac_energy_PumpandStir;
% Electric use pump and stir (MJ/ha)
El_Compression = Prod_upgrad_biogas* Fac_energy_compression;
% Electric use compression (MJ/ha)

Fac_energy_Compaction = 0.343;   % L diesel/Mg DM biomass, compaction of
                                 biomass in silo (Prade et al., 2015)
Frac_energy_Upgrad = 0.03;       % fraction of produced energy (Bauer et
                                 al., 2013)
Frac_energy_PumpandStir = 0.054;% MJ/kg feedstock (Björnsson et al., 2013)
Frac_energy_compression = 0.026; % fraction of produced energy, compression
                                 of biogas to 200 bar (Björnsson et al.,
                                 2013)
```

2.3 CH4 emissions anaerobic digestion

```
CH4_loss_BioGas_prod = Prod_upgrad_bio-
gas/Energy_cont_bio*CH4_dens*(CH4_loss_ana + CH4_loss_upgrad);

% CH4_losses_BioGas_prod = kg CH4 losses during digestion and upgrading
CH4_dens = 0.83;                 % Density CH4 kg/m3 (Gode et al., 2011)
CH4_loss_ana = 0.0001;           % Fraction lost CH4 in digestion per CH4
                                 produced (Uppsala vatten, 2017)
CH4_loss_upgad = 0.003;          % Fraction lost CH4 in upgrading per CH4
                                 produced (Uppsala vatten, 2017)
```

2.4 Emission digestate storage

```
NH3_loss_DigStor = yield*(1 - DM_loss_ens)*N_cont_grass*NH4_frac_dig
*NH3_Fac;
% NH3_loss_DigStor = NH3-N losses during digestate storage
CH4_loss_DigStor = Digestate*(1 - DM_loss_ens)*VS*B0*0.67*MCF;
% CH4_loss_DigStor = CH4 losses during digestate storage, 0.67 is the con-
version factor between volume and weight of CH4, is derived at 101 kPa and
19°C, Digestate is the VS left in the biomass after the digestion
N2O_loss_DigStor = NH3_loss_DigStor*0.01;
% N2O_loss_DigStor = N2O-N losses during digestate storage

N_cont_grass = 0.022;            % Nitrogen content in grass silage kg N/kg
                                 DM (Styles et al., 2015)
NH4_frac_dig = 0.60;             % Fraction of NH4 of total N in digestate
                                 (Tidåker et al., 2016)
NH3_Fac = 0.1;                   % Factor for open digestate storage tank
                                 (Misselbrook et al., 2012; Styles et al.,
                                 2016)
```

```
B0 = 130;                          % The CH4 that can theoretically be ex-
                                   tracted from the VS (Björnsson et al.,
                                   2013)
MCF = 0.035                        % Methane conversion factor, country spe-
                                   cific value for liquid manure(Swedish EPA,
                                   2018)
```

3 Eq S3: Soil fertilisation (ΔSF)

```
C_in_dig = yield*C_in_DM - (Prod_upgrad_BioGas/CH4_cont_biogas)-Prod_up-
grad_BioGas)*co2_to_C - (Prod_upgrad_BioGas + CH4_loss_BioGas_prod +
CH4_loss_DigStor)*CH4_to_C;
% C in the digestate (Mg/ha)
N_in_dig = ((yield*(1 - DM_loss_ens)*N_cont_grass) - NH3_loss_DigStor –
N2O_loss_Digstor;
% N in the digestate (Mg/ha)
CtoN_dig = C_in_dig/N_in_dig;
% C:N ratio in the digestate

C_in_DM = 0.4;                     % C fraction in DM
CH4_cont_biogas = 0.55;            % CH4 content in biogas before upgrading
                                   (Edström et al., 2008)
Co2_to_C = 12/44;                  % Conversion factor CO2 to C
Ch4_to_C = 12/16;                  % Conversion factor CH4 to C
```

4 Diesel

```
Diesel_CO2 = (5.78 + 73)/1000;     % kg CO2/MJ, production + utilisation
Diesel_CH4 = (0.0338 +
0.00054)/1000;                     % kg CH4/MJ production + utilisation
Diesel_N2O = (0.0000555 +
0.002)/1000;                       % kg N2O/MJ production + utilisation

EF_Diesel = Diesel_CO2 + (Diesel_CH4 * CF_CH4) +
(Diesel_N2O * CF_N2O);
% emission factor Diesel kg CO2 eq/MJ

Calorific_Value_Diesel = 35.3;     % MJ/L (Gode et al., 2011)
PEF_Diesel = 1.09;                 % Primary energy factor MJ/MJ (Gode et
                                   al., 2011)
```

5 Nordic electricity mix

```
EF_NordElmix = 36.5;               % g CO2eg/MJ (Martinsson et al., 2012)
PEF_NordElmix = 1.74               % Primary energy factor MJ/MJ
```

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