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.

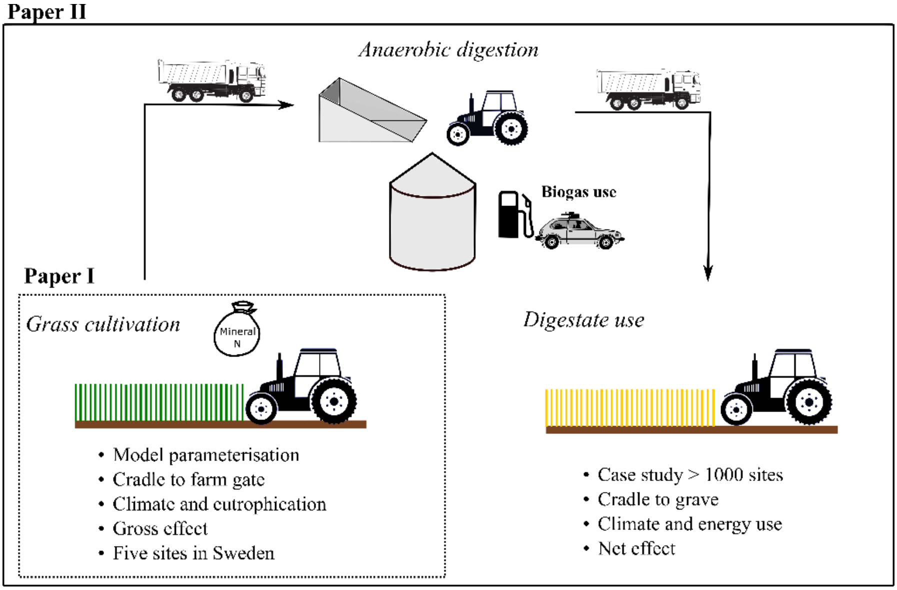

Figure 1. Schematic illustration of the work reported in Papers I and II and the links between the papers.

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*).

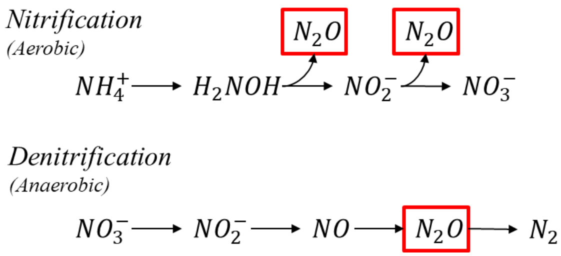

Figure 2. Production and consumption of the different reactants in nitrification and denitrification.

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.

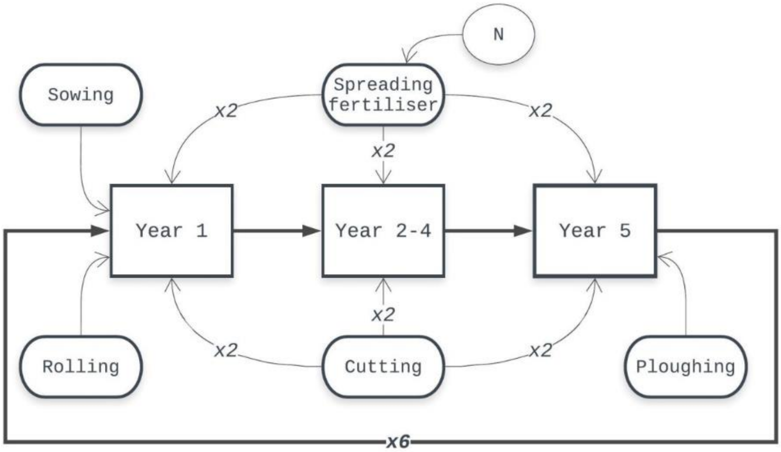

Figure 4. Schematic overview of the grass cultivation rotation analysed at the five sites in Paper I. The grass was sown and the soil was rolled in year 1 and the grass was terminated with ploughing to 30 cm in year 5. During the crop rotation, the grass was fertilised and cut twice a year.

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.

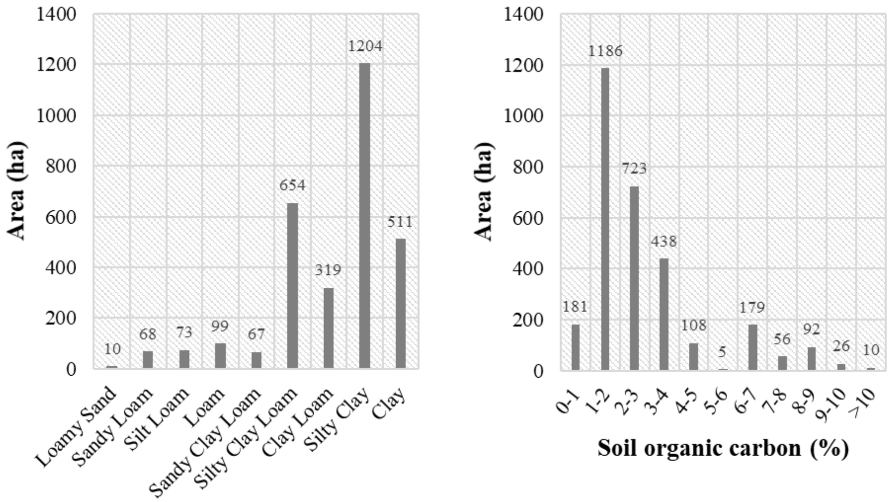

Figure 5. Soil texture and initial soil organic carbon (SOC) content at sites under fallow (N=1240) in Uppsala municipality used in Paper II.

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.

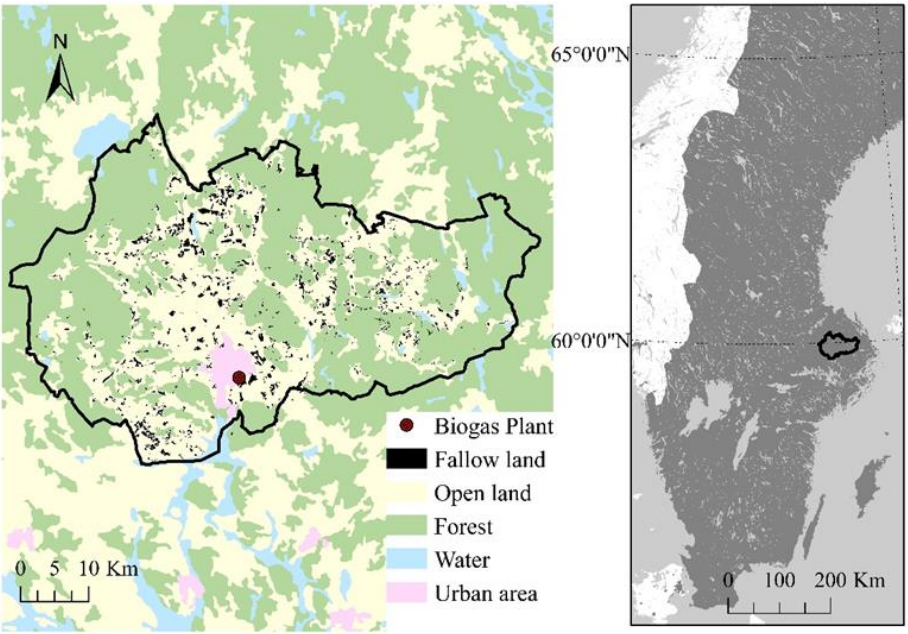

Figure 6. (Left) Map of the study region of Uppsala municipality (inside the black line), showing the distribution of fallow land (black dots) and the location of the municipal biogas plant (red and black dot). (Right) Map of Sweden showing the location of Uppsala municipality.

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:

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

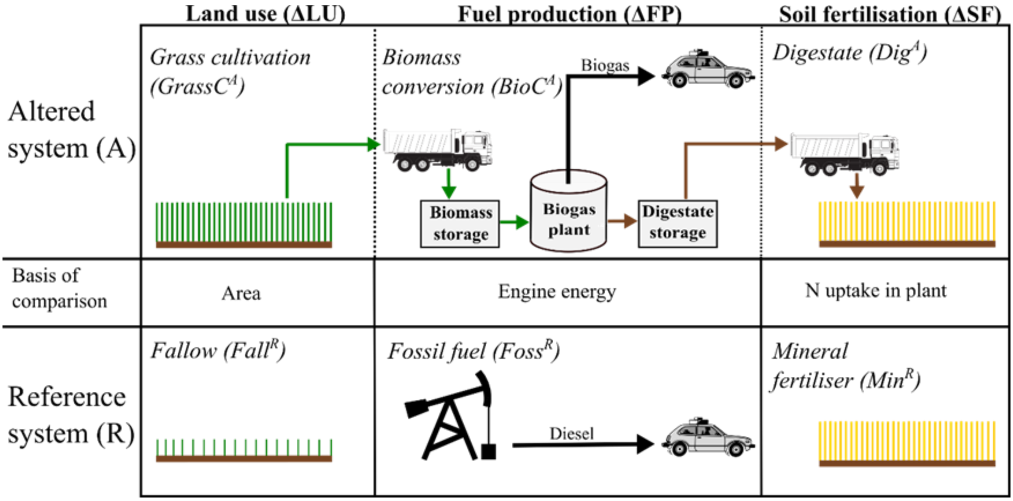

Figure 7. Schematic illustration of the grass-based biogas system studied in Paper II, divided into six subsystems: Grass cultivation (*GrassC*^{A}), Biomass conversion (*BioC*^{A}), Digestate use (*Dig*^{A}), Fallow (*Fall*^{R}), Fossil fuel (*Foss*^{R}) and Mineral fertiliser (*MinR*). The net effect 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:

$$Energy\,ratio = E_{OUT}/E_{IN} \tag{4}$$

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.

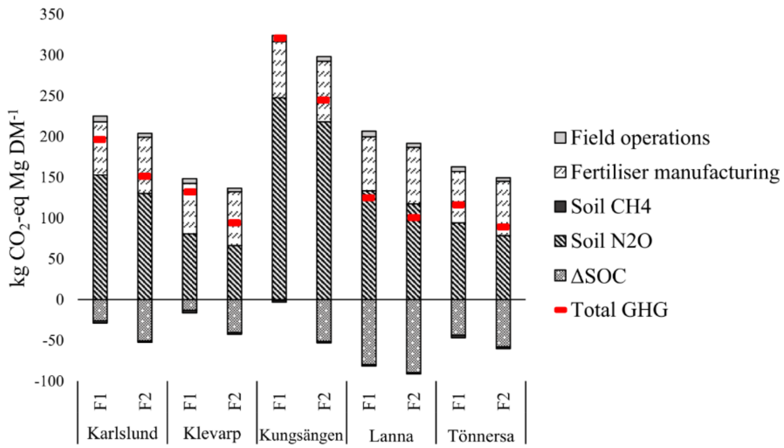

Figure 8. Total climate impact of grass cultivation during 30 years at the five study sites in Paper I for fertilisation rates F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}), assessed as Global Warming Potential over 100 years (GWP_{100}).

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.

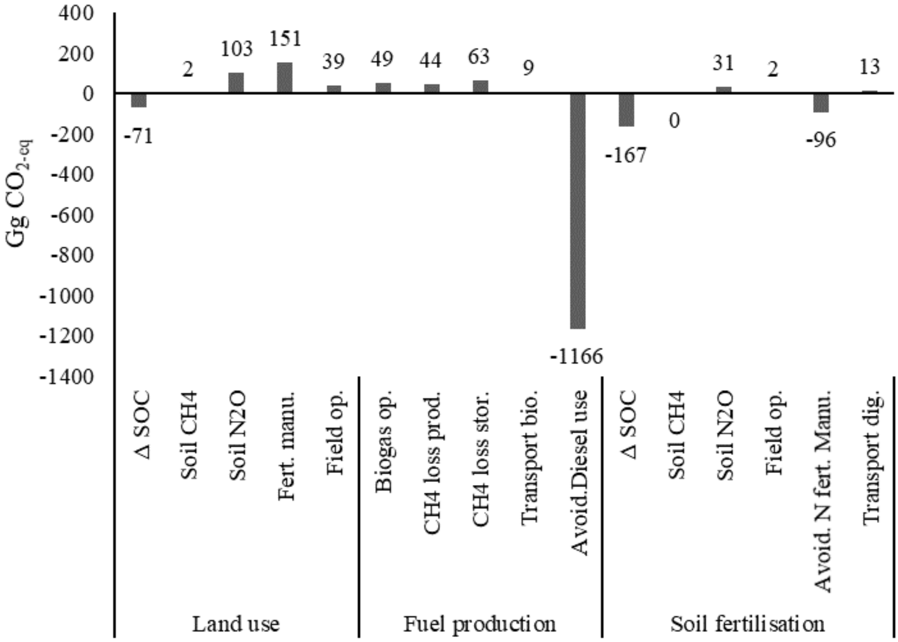

Figure 9. Total net climate impact of the grass-based biogas system in Uppsala municipality over 100 simulated years. The emissions (assessed as GWP_{100}) are aggregated into the three compartments: Land use, Fuel Production and Soil fertilisation, and are expressed in Gg CO_2eq (G = 10^9)

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.

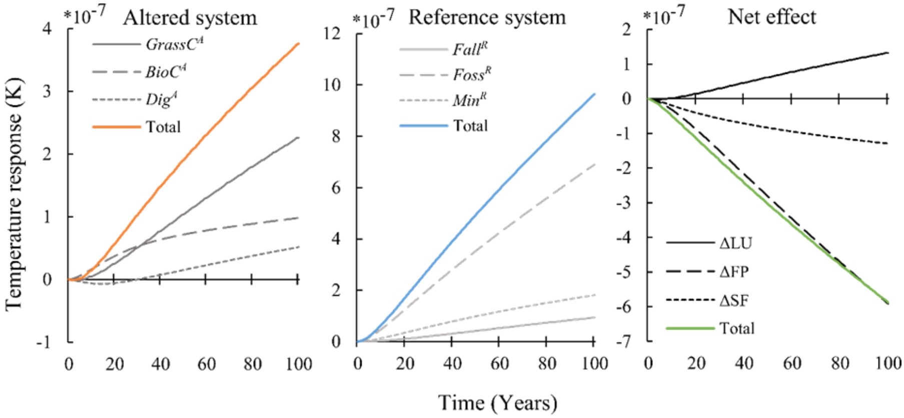

Figure 10. Temperature response, in degrees Kelvin (K) using all fields studied (N=1240, 3006 ha) in the region, for (left) the altered system and (centre) the reference system, and (right) the total net effect.

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%.

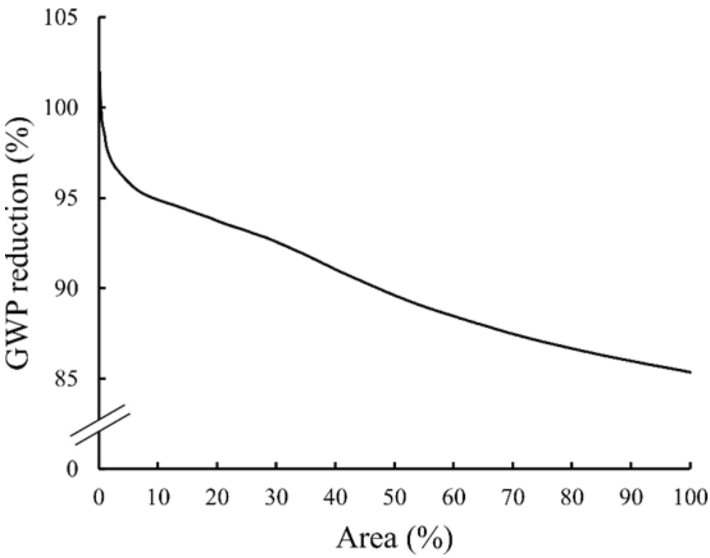

Figure 11. Global warming potential (GWP) reduction compared with diesel from using the grass-based biogas system, without fossil fuel substitution (*FossR*), in relation to fraction of total available land area used.

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).

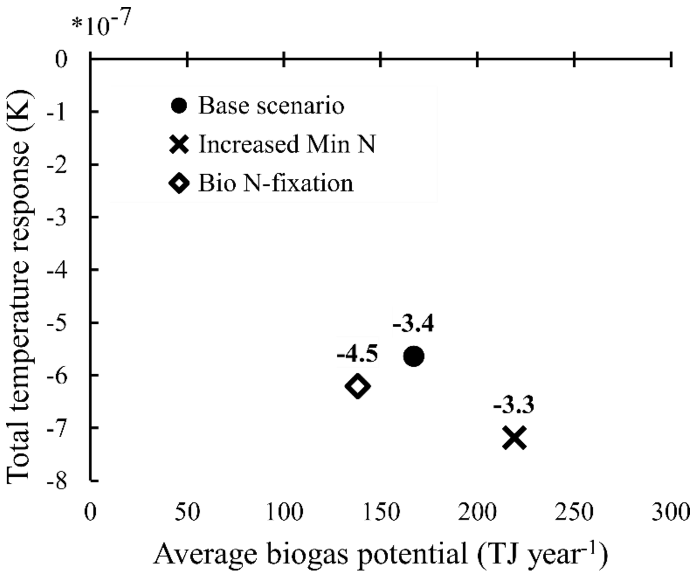

Figure 12. Total temperature response (degrees K), over 100 years, and average biogas potential (TJ per year) for the base scenario and for two alternative scenarios: increased mineral nitrogen (Min N) fertilisation and use of biological N-fixing crops. The values next to the points show the temperature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

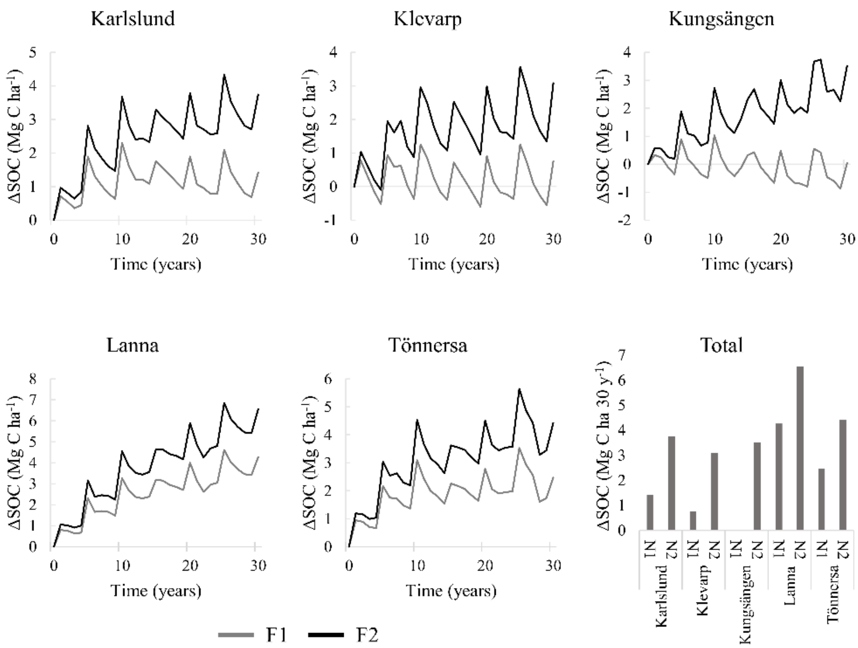

Figure 13. Simulated cumulative change in carbon (C) stock for the five sites investigated in Paper I and the total for all sites. The grey line represents fertilisation rate F1 (140 kg N ha^{-1}) and the black line F2 (200 kg N ha^{-1}).

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.

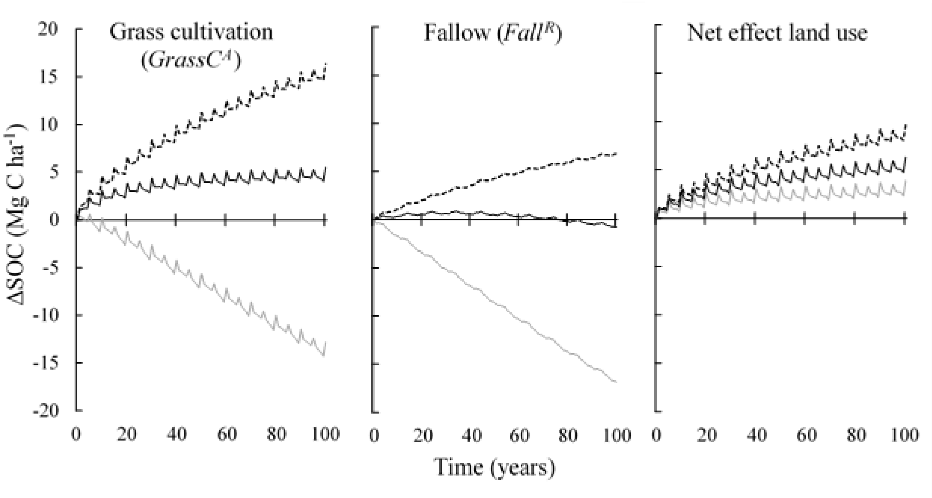

Figure 14. Simulated cumulative change in soil carbon (C) sequestration in the study region of Uppsala municipality. Change in C stock (Mg C ha^{-1}) for (left) grass cultivation only and (centre) fallow only, and (right) net effect of the land use, *i.e*. grass cultivation – fallow. The grey line represents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

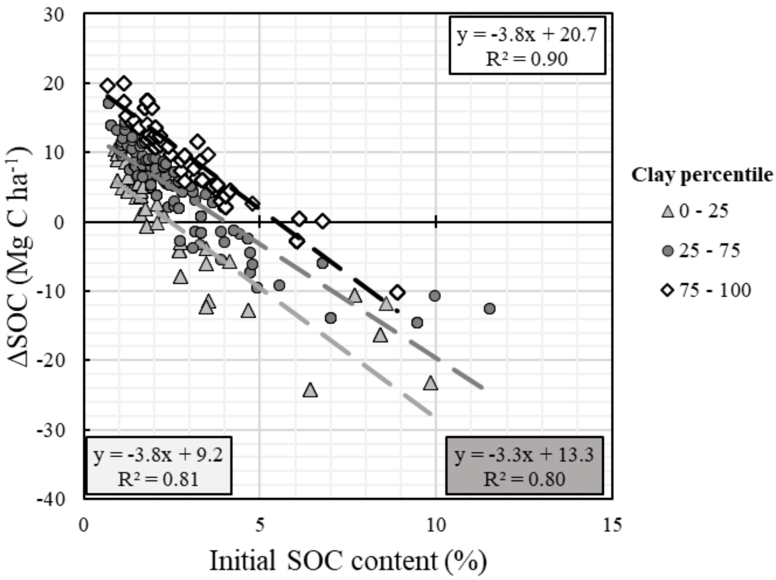

Figure 15. Change in soil carbon (C) content (%) after 100 years of grass cultivation, related to the input parameters with the largest correlation for the study region, initial C content and clay content. Total C change from grass cultivation on the y-axis, initial C content on the x-axis. The colour 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).

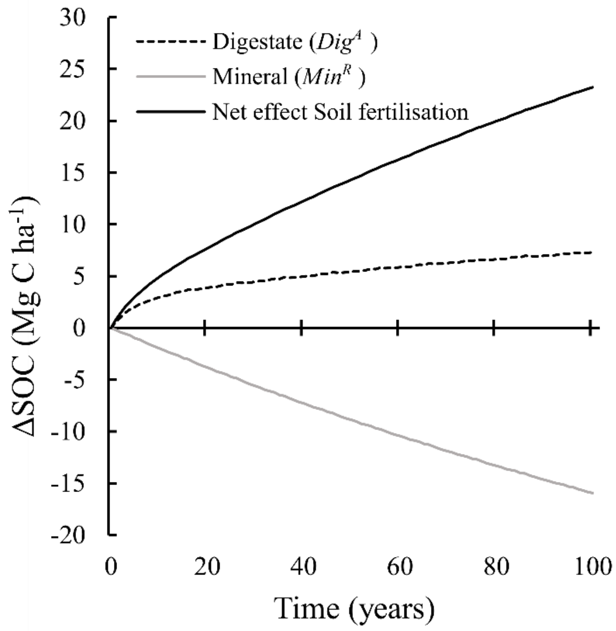

Figure 16. Simulated soil carbon (C) sequestration in winter wheat cultivation using mineral fertiliser (grey), digestate fertiliser (dashed) and the net effect, *i.e.* digestate – mineral. Modelled for one field, which represented average conditions in the study region

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).

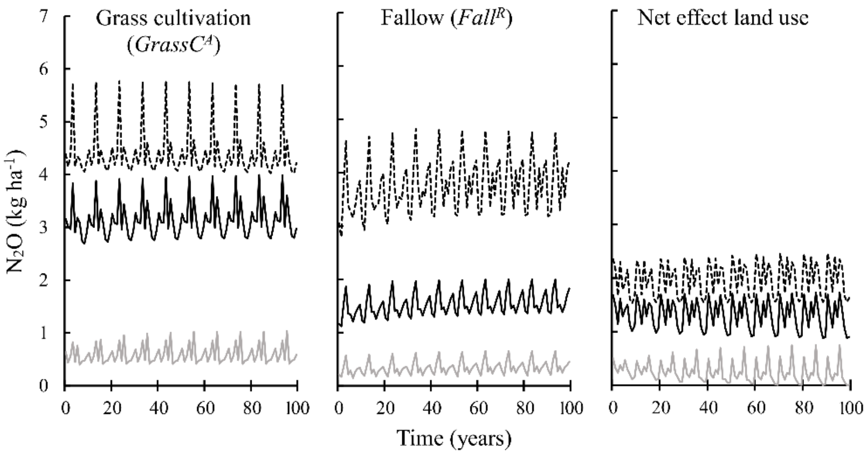

Figure 17. Annual nitrous oxide (N_2O) emissions from soil in (left) the grass system and (centre) from fallow land, and (right) net emissions from biogas feedstock cultivation during 100 years for all sites (N=1240). The dashed black line represents the 95th percentile soil (max), the grey line the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

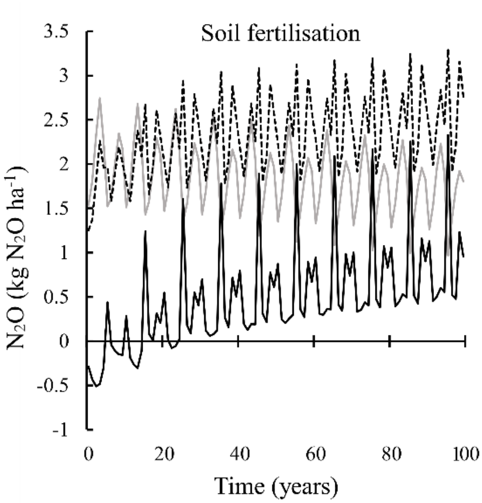

Figure 18. Annual nitrous oxide (N_2O) emissions from soil over a 100-year time horizon following fertilisation of winter wheat. The grey line represents emissions from winter wheat cultivation with mineral fertiliser, the black dashed line emissions from winter wheat cultivation with digestate. The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

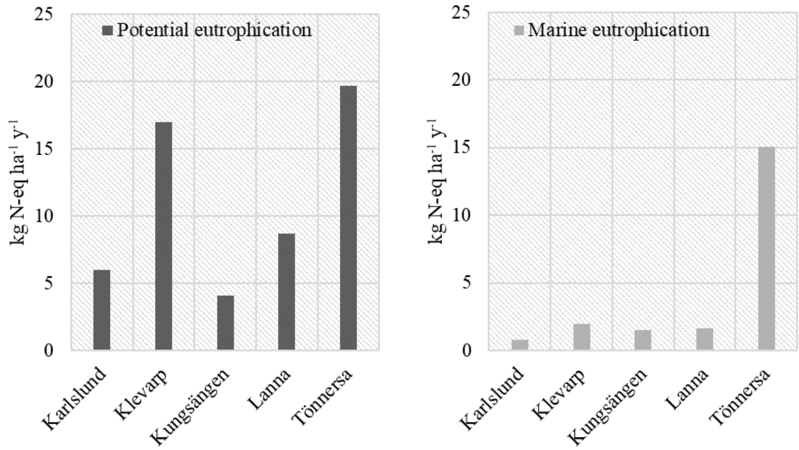

Figure 19. (Left) Potential eutrophication impact assessed using the CML approach and (right) marine eutrophication impact assessed using the methodology of Henryson *et al*. (2018)

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

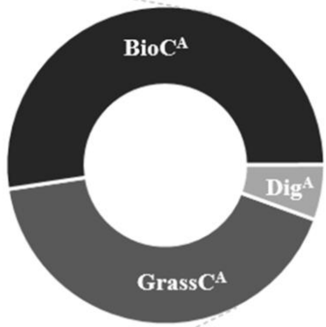

Figure 20. Annual primary energy (PE) input and energy output of the altered system for the study region of Uppsala municipality, divided between the subsystems grass cultivation (*GrassC^A*), biomass conversion (*BioC^A*) and digestate use in winter wheat cultivation (*Dig^A*).

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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