GRASS FOR BIOGAS – ARABLE LAND AS A CARBON SINK

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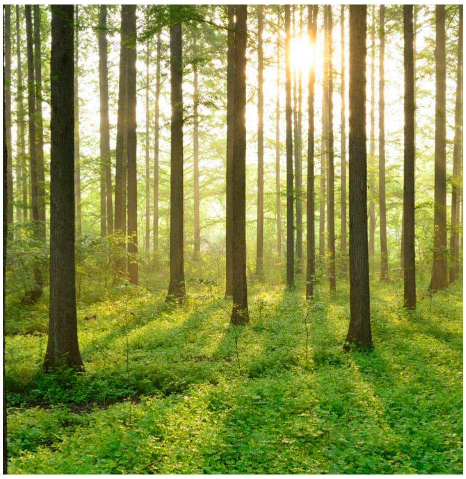

TRANSPORTATION AND FUELS

Energiforsk

Grass for biogas – Arable land as a carbon sink

An environmental and economic assessment of carbon sequestration in arable land through introduction of grass for biogas production

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Foreword

In this report, the results from the research project *Sustainability for ley as biogas crop – climate, land use and costs* are presented. The researchers behind the project are Lovisa Björnsson, Thomas Prade and Mikael Lantz at Environmental and Energy Systems Studies, Lund University. Lovisa, professor of environmental biotechnology and bioenergy, has acted project leader and has had main responsibility for assessment of environmental impact and modelling of grass properties and biogas production. Thomas has performed this project as part of his post doc period at Lund University, and is now working as a researcher at Biosystems and Technology, Swedish University of Agricultural Sciences, Alnarp. He has had main responsibility for the soil carbon modelling and for the outline of crop rotations, inputs and cost assessments in cultivation. Thomas and Lovisa also together collected and analysed data and statistics for the selection of regions. Mikael, assistant university lecturer, has had main responsibility for cost assessments, energy inputs and emissions in biogas production. The project has been running 2014-2016. An extended summary in Swedish can be found at http://miljo.lth.se/publikationer/forskningspublikationer/ with the title:

Lovisa Björnsson, Thomas Prade & Mikael Lantz (2016) Åkermark som kolsänka – en utvärdering av miljö- och kostnadseffekter av att inkludera gräsvall för biogas i spannmålsrika växtföljder. Rapport Nr 98, Miljö- och energisystem, Lunds Universitet. ISBN 978-91-86961-24-4.

Sammanfattning

På grund av ökande specialisering, intensifiering och minskad användning av biogödsel har vi i Sverige idag regioner där vi tappar organiskt material i åkermarken. Två regioner i Skåne och Västra Götaland som karaktäriseras av spannmålsdominerade växtföljder och låg djurtäthet valdes ut, och beräkningarna visade att vi här i nuläget tappar kol från åkermarken i en omfattning som bidrar till ett utsläpp av koldioxid som i Västra Götalandsområdet är nästan 4 gånger så stort som växthusgasbidraget från dieselanvändningen i odling. För att vända denna utveckling krävs en ökad koltillförsel, vilket kan fås t ex genom högre tillförsel av odlingsrester eller genom biogödsling. I de alternativa framtidsscenarier som analyserades förändrades växtföljderna till att inkludera gräsvall under 2 av 6 år. I dessa regioner finns dock liten avsättning för gräs som djurfoder, och gräset antogs istället användas som energigröda för biogasproduktion. Detta val gjordes också för att illustrera konflikten i när energigrödor ersätter livsmedels/fodergrödor på åkermark, och demonstrera vilka för- och nackdelar det kan innebära. Den producerade biogasen antogs användas som drivmedel för tung trafik och producerad biogödsel användes i odlingen där den ersatte mineralgödsel.

Markkolsutvecklingen i dessa alternativa scenarier kunde vändas, i Skåne till att göra åkermarken till en kolsänka, i Västra Götaland till att dagens markkolshalt kunde bibehållas. Att introducera gräs i spannmålsväxtföljden i dessa regioner, 274 000 ha, skulle kunna bidra med 1,9 TWh biogas, vilket är mer än hela dagens produktion i landets biogasanläggningar (1,6 TWh 2014). Klimatnyttan i odlingsledet skulle motsvara 0,2 miljoner ton CO_2-ekvivalenter per år, och användningen av biogasen som ersättning för diesel skulle ge en ytterligare lika stor utsläppsreduktion. Samtidigt som hållbarheten ur markkvalitetsperspektiv skulle öka skulle dock spannmålsproduktionen minska med 270 000 ton (varav 2/3 vårkorn) per år, vilket motsvarar drygt 10 % av dagens spannmålsanvändning som djurfoder, och med drygt 100 000 ton rapsfrö, vilket motsvarar 10 % av dagens svenska rapsbaserade biodieselanvändning.

Den livscykelanalys (LCA) som genomfördes visade även på andra förändringar i miljöpåverkan, som att partikelemissionerna skulle öka i produktionen, men minska då biogasen ersatte diesel och ge en minskning totalt sett. Introduktionen av gräs i växtföljderna minskade kväveläckaget, men biogödslingen ökade emissionerna av ammoniak till luft, vilket gav ett ökat bidrag till både övergödning och försurning. Förändringen är alltså inte entydigt positiv, vilket visar hur viktigt det är att anlägga ett brett perspektiv för att utvärdera miljöpåverkan. Det sammantagna samhällsekonomiska värdet av förändringen uppgick till 1 500 – 2 500 kr per ha (per ha i hela den studerade växtföljden) trots det negativa bidraget till övergödning och försurning.

I de undersökta regionerna kan ett miljöstöd om 500 kr per ha erhållas för vallodling, där syftet är att stimulera hållbar odling och minska läckaget av växtnäring, men vall odlas trots detta enbart på en mindre del av åkermarken. Vi undersökte vilket gräspris som skulle krävas för bibehållen intäkt jämfört med nuvarande växtföljd. Detta pris användes för att beräkna kostnaden för producerad biogas i en nybyggd anläggning med gräs som enda råvara. Med nuvarande gaspriser skulle gräs under dessa förutsättningar vara för dyr som biogasråvara, det skulle krävas en minskning av råvarupriset med 20 % eller en ökning av gaspriset med 6 % för att få en lönsam

produktion ur biogasperspektiv. En förutsättning skulle också vara att produktionskedjan uppfyller kraven på 60 % reduktion av växthusgasemissioner som gäller för nya anläggningar fr o m 2015. Detta är fram till 2020 en förutsättning för att biogasen ska befrias från CO_2-skatt i Sverige, och utan denna fördel har inte biogasen någon möjlighet att konkurrera med de fossila alternativen. Växthusgasreduktionen ska då beräknas i enlighet med metodiken i EUs förnybartdirektiv (EU RED). Där exkluderas dock vissa aspekter, som t ex markkolsnyttan av att inkludera gräs i växtföljden. Klimatnyttan enligt denna beräkning framstår därför som betydligt sämre än vid den LCA-baserade beräkningen, och gräs till biogas kunde bara med nöd och näppe klara kravet på 60 % reduktion.

Som en jämförelse gjordes även denna beräkning för en djurrik region i Småland som redan har vallodling på mer än 80 % av åkermarken. Här beräknades ett alternativscenario baserat på intensifiering av nuvarande vallodling och på att gräset skulle tillföras en befintlig biogasanläggning med gödsel som huvudråvara. Detta alternativ var under antagna förutsättningar både ekonomiskt gångbart och kunde uppfylla kraven på växthusgasreduktion. I denna region är inte markkolseffekten stor, och heller inte ett problem vid nuvarande åkermarksanvändning. Detta illustrerar hur markkolsvärdet varken värderas vid en ekonomisk bedömning eller tydliggörs när klimatnyttan beräknas enligt EU RED. Det blir därför inte en aspekt med betydelse i en hållbarhetsbedömning baserad på enbart dessa kriterier. I EU RED diskuteras aspekter som indirekt påverkan på markanvändning (iLUC), och att det kan kullkasta hela klimatnyttan när vissa livsmedelsgrödor används för biodrivmedelsproduktion, men den stora klimatpåverkan som en markkolsförlust innebär exkluderas.

Det övergripande syftet med denna studie har varit att ta fram fakta som ökar förståelsen för det breda perspektiv som krävs för beslut kring långsiktigt hållbar åkermarksanvändning. Förlusten av markkol från åkermark är inte hållbar på lång sikt, och åtgärder måste förr eller senare vidtas för att bryta denna utveckling. En hållbar användning av åkermark ska ge lägsta möjliga bidrag till växthusgasemissioner samtidigt som livsmedelsproduktionen säkras på lång sikt. Att införa odling av gräsvall i spannmålsväxtföljder enligt det alternativ som har studerats här skulle stoppa eller till och med vända den kolförlust från åkermark som sker i dagens odling och bidra till minskade växthusgasutsläpp, både i odling och inom transportsektorn. Det kan vara samhällsekonomiskt motiverat att uppmuntra denna förändring även om den innebar negativ påverkan på andra miljöaspekter.

Det är viktigt att vidga perspektivet och göra tillräckligt omfattande analyser av så komplexa system som användning av åkermark. Att väga in lokala förutsättningar, titta på odlingssystem och växtföljdseffekter istället för på enskilda grödor och på konflikten mellan olika miljömål är de perspektiv som studerades här. Hållbarhetskriterierna i dagens förnybartdirektiv inom EU omfattar inte de olikheter i förutsättningar och landskapsperspektiv som finns i unionen. Den policy som nu formuleras i EU inför 2020 vad gäller bioenergi bör istället utformas som ett övergripande ramverk, och hållbarhetskriterier där det är viktigt att ta hänsyn till lokala förutsättningar och rumsliga perspektiv formuleras förslagsvis baserat på vetenskapligt välgrundade bedömningar på nationell nivå.

Summary

Due to increasing specialization, intensification and reduced use of bio-fertilizer, we have in Sweden today regions where we lose organic matter in arable land. Two regions in Skåne and Västra Götaland, characterized by cereal-dominated crop rotations and low animal density, were selected for the evaluation. The calculations showed that we currently lose carbon from arable land in the Västra Götaland region to an extent that contributes almost four times the equivalent of the greenhouse gas contribution from diesel use in farming. To reverse this trend, an increased supply of carbon is required, which can be obtained for example through higher supply of crop residues or by applying bio-fertilizer. In the alternate future scenarios analyzed, the crop rotations were changed to include grass in 2 of 6 years. In these regions, there is little demand for grass for animal feed, and the grass was instead used as an energy crop for biogas production. This choice was also made to illustrate the conflict as energy crops replace food / feed crops on arable land, and the advantages and disadvantages it may entail. The biogas produced was adopted use as fuel replacing diesel, and the produced bio-fertilizer was used in cultivation where it replaced mineral fertilizer.

The soil carbon development in these alternative scenarios could be reversed, in Skåne to make farmland a carbon sink, in Västra Götaland so that today's soil carbon content could be maintained. To introduce grass in cereal crop rotation in these regions, 274,000 hectares, could contribute with 6.8 PJ per year of biogas, which is more than the entire current production in the country's biogas plants (5.6 PJ in 2014). The climate benefit in cultivation would be equivalent to 0.2 million t CO2-equivalents per year, and the use of biogas as a replacement for diesel would a similar emission reduction in addition. While the sustainability of arable land use was improved from a soil quality perspective, the grain production fell by 270 000 t (of which 2/3 spring barley) per year, equivalent to just over 10 % of today's cereal use for animal feed, and by more than 100 000 t of rapeseed, which is equivalent to 10 % of today's Swedish rapeseed-based biodiesel use.

The life cycle assessment (LCA) also revealed other environmental impacts, such as that particle emissions would increase in production, but decrease as biogas replaced diesel and provide an overall decrease. The introduction of grass in the crop rotations reduced nitrogen leaching, but the biofertilizer utilization increased emissions of ammonia to air, resulting in an increased contribution to both eutrophication and acidification. The investigated change is thus not unambiguously positive, which shows the importance of a broad perspective in the evaluation of environmental impacts. The total socioeconomic value of the change amounted to 160 – 260 € per ha (per ha throughout the studied crop rotation), despite the negative contribution to eutrophication and acidification.

In the surveyed regions, a support for environmental measures of 50 € per ha can be obtained for forage cultivation, where the aim is to stimulate sustainable crop cultivation and reduce leaching of nutrients. Forage is, despite this, only cultivated on a small portion of the arable land. We investigated which grass price that would be required for sustained revenue compared to the current crop rotation. This price was used to calculate the cost of the biogas produced in a newly built facility with grass as the only raw material. At current gas prices, grass would under these conditions be too expensive as biogas feedstock, a reduction in feedstock price of 20%, or an increase in

gas price by 6% would be required to obtain a profitable production from biogas perspective. One condition would be that the production chain meets the demand on 60% reduction of greenhouse gas emissions that apply to new installations from 2015. This a prerequisite for biogas to be exempt from the CO_2 tax in Sweden until 2020, and without this advantage, the biogas cannot compete with fossil fuel alternatives. Greenhouse gas reduction should be calculated according to the methodology in the EU renewable energy directive (EU RED). In this method, however, certain aspects, such as the soil carbon impacts of including grass in the crop rotation, are excluded. The climate benefits when using the EU RED methodology are thus much lower than at the LCA-based calculation, and grass for biogas could just barely meet the requirement of 60% reduction.

As a comparison, calculations was also made for a region in Småland with high cattle density and grass cultivation on more than 80% of the arable land. Here, the evaluated modification was based on intensification of current grass production, and that the excess grass would be added to an existing biogas plant with cattle manure as the main feedstock. This option was under assumed conditions both economically viable and able to meet the EU RED requirements for greenhouse gas reduction. In this region, the soil carbon impact was small, and loss of soil carbon is not an issue at current arable land use. This illustrates how the value of the soil carbon impact is neither valued at an economic assessment or when the climate benefits are calculated according to the EU RED. It is therefore not an aspect of importance in the sustainability assessment based on these criteria alone. The EU RED takes the indirect impacts of land use (iLUC) into account, and that it can derail the entire climate benefit when certain food crops are used for biofuel production, but the big impact on the climate connected to soil carbon losses are excluded.

The overall aim of this study was to produce data to improve understanding of the broad perspective needed for decisions on sustainable use of arable land. The loss of soil carbon is not sustainable in the long term, and measures must sooner or later be taken to reverse this trend. A sustainable use of arable land should give the lowest possible contribution to greenhouse gas emissions while food production is safeguarded in the long term. To introduce grass cultivation in cereal crop rotations according to the modified scenarios that have been studied here would halt or even reverse the present carbon loss from arable land. This would contribute to reduced greenhouse gas emissions, both in crop cultivation and in the transport sector. It may be socioeconomically justified to encourage this change, even if it meant a negative impact on some environmental aspects.

It is important to broaden the perspective and to make sufficiently wide-ranging analyzes of such complex systems as the use of arable land. To take local conditions into consideration, to look at the effects of cultivation system and crop rotation instead of on individual crops and at the conflict between environmental objectives are the perspectives that were studied here. Sustainability criteria in the present EU renewable energy directive are not formulated taking local conditions and landscape perspectives into consideration. At present, the work on the EU bioenergy policy post 2020 is ongoing. To avoid contra productive measures, it seems important that future policies are formulated on a more broad level, and that sustainability criteria, where it is important to take local conditions and spatial perspectives into account, will be based on scientifically sound assessments at the national level.

List of content

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1 Acknowledgements

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2 Introduction

In the transition to a biobased society, we face the challenge of how to supply biomass in quantities large enough to replace fossil products in competition with the demand for food, feed and biomass-based materials. A major part of the biomass resources used in new applications in a biobased society will in the near future, just as the fossil resources today, be used for energy supply in different forms. While Sweden has a beneficial situation regarding the supply of renewable energy in general, a challenge is the transition of the transport sector. The demand for biofuels in transport has been predicted to correspond to, also in long-term predictions, around 70 PJ a^{-1} (SOU, 2013). In 2014, the EU average share of renewable energy in transport fuel consumption was 5.9% (Eurostat, 2016). The use of arable land for the production of biomass for biofuel production is the main pathway for the present biofuel supply within EU, with biodiesel representing around 80%. The EU biodiesel production in 2011 amounted to 340 PJ, requiring 20% of the world´s traded vegetable oil (Marelli *et al.*, 2015). This has led to an increasing debate about the conflicting use of available resources of arable land for energy crop production. Concerns about future food shortage has resulted in restrictions in the EU on the share of biofuels based on crops grown primarily for energy purposes on arable land, which should constitute no more than 7% of the final consumption of energy in transport in the member states after 2020 (EU, 2015). In Sweden, biofuels contributed with 12.6% of the domestic transport fuels in 2014, were 7.4% where produced based on crops from arable land (SEA, 2015a, b, c). We have the advantage of a more diverse biofuel supply, applying both low and high blend liquid biofuels replacing diesel and petrol, and using biogas as biofuel, an approach that opens up for a wider range of biofuel feedstocks.

With the aim of addressing the food-fuel conflict, the EU has introduced the so called indirect land-use change (iLUC) factors, which estimate indirect land-use change emissions from production of biofuel and bioliquid feedstocks. The iLUC factors add emission penalties on biofuels from starch rich crops, oil crops and sugar crops corresponding to the greenhouse gas emissions assumed to occur elsewhere if these crops are used for fuel and thus impacting the food crop market (EU, 2015).

Agriculture in Sweden has undergone radical changes and rationalizations in the last decades. Around 20% of the arable land that was cultivated in 1950 is no longer farmed (SEPA, 2015). A recent analysis shows that an area corresponding to 3% of the active arable land can be identified as abandoned, and that the decrease in cultivated arable land since 1999 has been 7% (Olofsson and Börjesson, 2016). At the same time, specialization, intensification and a decreased use of biofertilizers on arable land has in some regions led to mineralization of soil organic matter, which results in a release of carbon dioxide (SEPA, 2015).

Thus, it can seem a bit awkward that while iLUC emissions are calculated by implementing complex economic models (Ahlgren and Börjesson, 2011), the direct impact on soil organic carbon (SOC) for different utilization options for arable land are not included in the calculation method stipulated in the EU renewable energy directive (EU, 2009, 2015). It is increasingly being stressed that for efficient climate mitigation, we must both look at replacing fossil fuels and look at the possible carbon sinks (e.g SEPA (2015)).

The aim of the present study was to present facts related to scenarios where crops from arable land are used for biofuel production. But instead of using starch rich, sugar or oil crops, grass was cultivated, and used for biofuel production. We believe there is a lack of facts to assess the full complexity and implications of such changes. As has been suggested by Hildingsson and Johansson (2016), policy measures that are designed to include several sustainability concerns need to take local contexts into account, and respond to new knowledge.

The assessment was made for Swedish regions with cereal-dominated crop rotations, where the introduction of grass in the crop rotation was believed to have a strong positive impact on SOC content. The scenarios were evaluated from a climate perspective, taking SOC changes into account. SOC changes were earlier shown to have an important role in greenhouse gas (GHG) mitigation in such systems (Björnsson *et al.*, 2013). A region with high livestock density, substantial grass production but minor cereal production was also evaluated for comparison.

Climate benefits were in focus of this study, but any change in the use of arable land will impact other sustainability aspects. Therefore, impact on eutrophication, acidification and particle emissions was also evaluated. Finally, the economic implications for the farmer, the ability for the biofuel producer to pay for grass as feedstock and the socioeconomic values were outlined and compared.

3 Methods

The study was based on a range of approaches and method applications which are summarized here, and further described in the respective sections.

- Inventory and processing of statistics and data from the Swedish Board of Agriculture (SJV), Statistics Sweden (SCB) and the Swedish Energy Agency (SEA).
- Inventory and processing of georeferenced data on soil properties for a GIS assessment of the study regions (Eriksson *et al.*, 2010).
- Modelling of the impact of crop rotation changes on soil organic carbon (SOC) content (Andrén and Kätterer, 1997).
- Assessment of environmental impacts (greenhouse gas emissions, eutrophication, acidification and particle emission) according to life cycle assessment methodology (ISO, 2006).
- Assessment of greenhouse gas emissions applying the methodology defined in the renewable energy directive (EU, 2009, 2015).
- Modelling of nitrogen leakage from arable land using the "VERA" programme from the Swedish Board of Agriculture (SJV, 2015a).
- Assessment of feedstock production costs using a stepwise calculation method considering field, transport and storage operations.
- Assessment of biogas production cost using investment analysis based on the annuity method.
- Applying literature-based socioeconomic values for environmental impacts to calculate socioeconomic impacts for the suggested modifications.

Chapters 4-8 contain the main features of scenarios and assessments, while all details on methods and the selection of data can be found in the appendix section. Results and conclusions can be found in Chapters 9-10.

4 Regions for assessment

This study was performed for two types of Swedish regions. The focus was on regions niched to crop/cereal production, with little animal husbandry and low availability of manure as biofertilizer. The expected characteristics for this type of region was that losses of soil organic matter were occurring with current crop rotations and cultivation practices. We also included a region characterized by livestock production, where grass (-clover) crop production already was common. The selection was made based on the present situation and historic development in agriculture, and details are presented in Appendix A.

Criteria for choosing the cereal type study regions were a) high share of cereal cultivation on arable land (>45%), b) low share of grass-clover crop cultivation on arable land (<40%) and c) low numbers of livestock expressed as livestock units per hectare (<0.3). Two cereal-dominated (C) regions were identified, C1 in the very south of Sweden, situated within the *Gss* (Götalands södra slättbygder) and *Gmb* (Götalands mellanbygder) production areas and C2, south of lake Vänern, situated mainly within the *Gns* (Götalands norra slättbygder) production region. This analysis is presented as share of cereals and grass crops of total arable land as average for the period 2003-2014 (Figure 1 and Figure 2), and the number of livestock units per hectare (ha) arable land (Figure 3). (The chosen regions consists of a number of harvest areas (skördeområde, SKO), which is the statistical basis for data on crop production (Figure 1 and Figure 2). For livestock production, statistical data was available on municipality level (Figure 3). Municipalities were chosen to represent the same areas as covered by the harvest areas. For more information, please refer to appendix A.

Criteria for choosing the livestock type of study region were a) low share of cereal cultivation on arable land (<15%), b) high share of grass crop cultivation on arable land (>60%) and c) high numbers of livestock expressed as animal units per hectare (>0.6). One region was selected as livestock region (L,) which was geographically close to the cereal regions C1 and C2, situated within the *Gsk* (Götalands skogsbygder) production area (Figure 1, Figure 2, Figure 3).

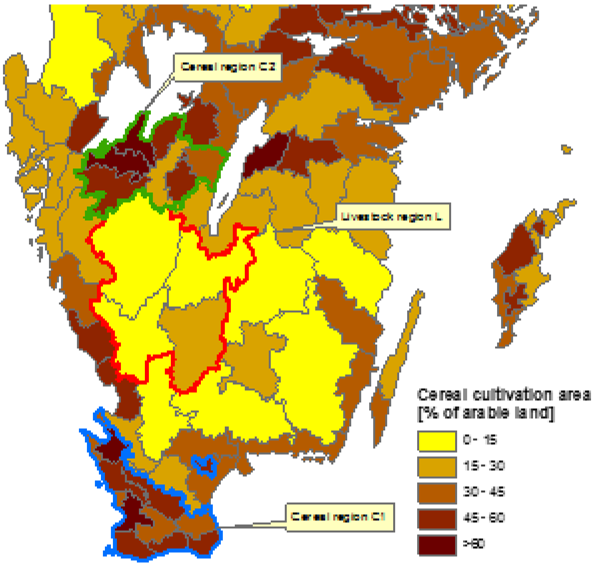

Figure 1. Average cultivation area of cereals (2009-2014) expressed as share of arable land in southern Sweden, based on data from Olsson (2015).

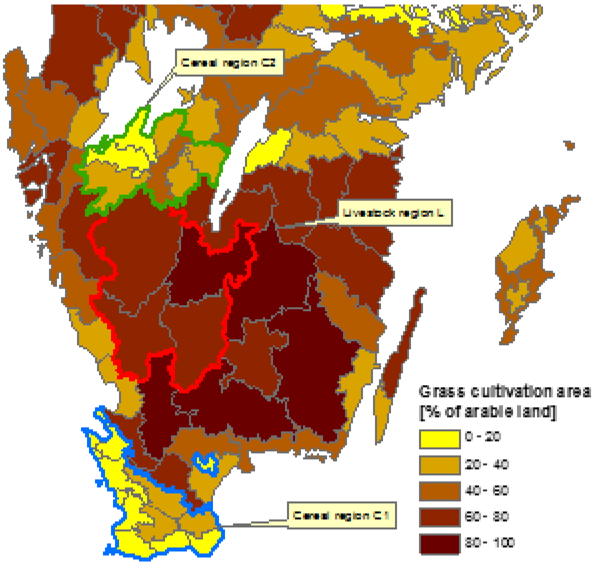

Figure 2. Average cultivation area of grass crops (2009-2014) expressed as share of arable land in southern Sweden, based on data from Olsson (2015).

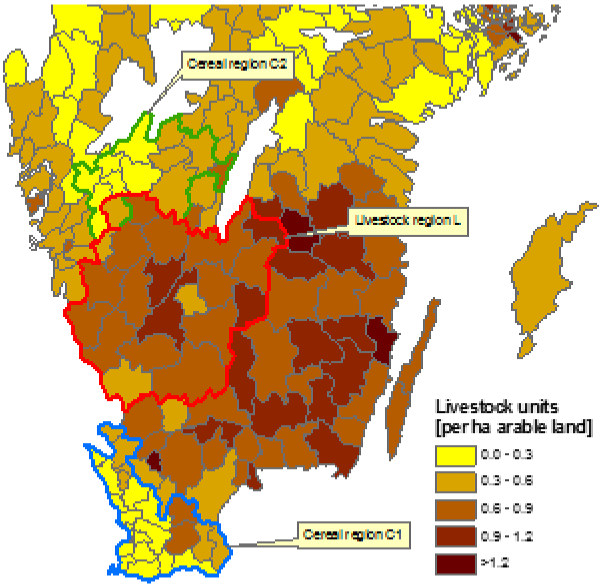

Figure 3. Average animal density (2009-2013) expressed as animal units per hectare arable land in southern Sweden, based on data from SJV (2015d).

5 Scenarios and evaluated aspects

Current crop production in the study regions was used as reference scenario (*current*). For each of the three selected study regions, an alternative crop production scenario was described (*modified*), leading to introduction of (C regions) or an intensification in (L region) grass production. For the C regions, a low demand for grass as cattle feed was assumed, and for the L region, the demand was assumed to be supplied by the current level of grass production. Thus, the grass produced in the modified scenarios was assumed to have no market as cattle feed, and therefore assumed to be used for the production of biogas. This biogas was upgraded to vehicle fuel quality, and the digestate (the effluent from grass digestion) was used as biofertilizer in the crop rotations, partly replacing mineral fertilizers. The scenarios and the aspects evaluated differ between the cereal regions and the livestock region, as described below.

5.1 CEREAL REGIONS

For the C regions, typical crop rotations reflecting the current crop production were defined and used as reference scenarios (*current*). In the modified scenarios, the crop rotations were changed to include grass production as further described in Chapter 6. The grass was transported to the biogas plant where it was used as biogas feedstock. As a sensitivity analysis on the impacts of scale of biogas production, two scales were evaluated per region as further described in Chapter 7. The general outline for the investigated C region scenarios are summarized in Table 1.

Table 1. General outline for the investigated current and modified scenarios in the cereal regions C1 and C2.

Scenario	current	modified
Short name	C1:c and C2:c	C1:m and C2:m
Crop rotation	Typical for the region	Modified to include grass
Biogas production	-	Grass used as biogas feedstock
Crop fertilization	Mineral fertilizer	Mineral fertilizer partly replaced by biofertilizer from grass digestion

The impact of the change from the current to the modified scenario was evaluated through the assessment of the following aspects;

- Soil organic carbon (SOC) development was modelled for all scenarios and compared between the current and the modified crop rotations, both with and without the use of digestate as biofertilizer.
- Land use impacts. The area of arable land in each region where the modified scenario could potentially be applied was quantified together with the potential biofuel production and loss of crop production
- Greenhouse gas emissions from a crop rotations perspective (per ha arable land in the crop rotation) for current and modified scenarios were evaluated, integrating SOC changes in the assessment. The analysis was also performed from a biofuel perspective (per MJ fuel), and compared to the outcomes of applying the EU RED calculation method and sustainability criteria.
- Other environmental impacts for the current and the modified scenarios were evaluated and presented as eutrophication potential, acidification potential and particle emissions

- Grass feedstock production cost were estimated from the farmer´s perspective in two ways; the production cost (€ per tonne) and the feedstock price required to maintain the same economic result for the whole crop rotation as in the current scenarios.
- Biogas production cost was calculated based on the feedstock price required to maintain the same economic result for the whole crop rotation. Based on current market price for biogas used as a vehicle fuel, the feedstock price required for break-even for the biogas producer was calculated as well.
- The total environmental impact was recalculated to socioeconomic values of the change from the current to the modified scenarios.

5.2 LIVESTOCK REGION

In the livestock study region, the current crop production is focused on the production of grass as coarse feed for cattle, and manure is used as biofertilizer in crop cultivation. There is little interest amongst local farmers in supplying grass for biogas feedstock since the current production is used as cattle feed, there is no need for grass to diversify the crop rotation and the arable land not in use presently is considered difficulty to use in a rational way [1, 2].

A crop rotation was defined based on current typical conditions, where the amount of cattle manure used as biofertilizer in cultivation was based on current livestock density as further described in Appendix A. The manure was in the current scenario assumed not to be used for the production of biogas (Chapter 7.3). In the modified scenario, the grass yield in the current crop rotation is assumed to be increased by intensification (Chapter 6). The cattle manure is together with this additional grass assumed to be used for the production of biogas. This biogas production is modelled based on an existing biogas plant in this region with manure as the main feedstock, as further described in Chapter 7.3. The digestate from the biogas plant is used as biofertilizer in crop production. The general outline for the investigated L region scenario is summarized in Table 2.

Table 2. General outline for the investigated current and modified scenario in the livestock region

Scenario	current	modified
Short name	L:c	L:m
Crop rotation	Typical for the region	Same as in the current scenario, but with intensified grass production
Biogas production	-	Grass used as biogas feedstock together with cattle manure
Crop fertilization	Manure and mineral fertilizer	Biofertilizer (from grass and manure digestion) and mineral fertilizer

The impact of the change from the current to the alternative scenario was evaluated through the assessment of the following aspects;

- Soil organic carbon (SOC) development was modelled for both scenarios and compared between the current manure-fertilized crop rotation and the intensified crop rotation with use of digestate as biofertilizer.

[1] Carlsson, Håkan. Göteborg Energi. Personal communication April 2015.
[2] Ola Hallin, HS Sjuhärad. Personal communication October 2015.

- Greenhouse gas emissions from a biofuel perspective (per MJ fuel) where calculated applying the EU RED calculation method and compared to present and future sustainability criteria.
- Grass feedstock production cost were estimated from the farmer´s perspective in two ways; the production cost (€ per tonne (t, 1 t = 1 Mg)) and the feedstock price required to maintain the same economic result for the whole crop rotation as in the current scenarios.

6 Crop production

6.1 CROP ROTATIONS

Crop rotations reflecting the current crop production situation in the three study regions are presented in Figure 4. In the modified crop rotations in the C regions, two years of grass are included in the crop rotations, replacing cereals or oil crops. In the L region, the current crop rotation is maintained, but is modified through an intensification of the grass production. See Appendix A for details.

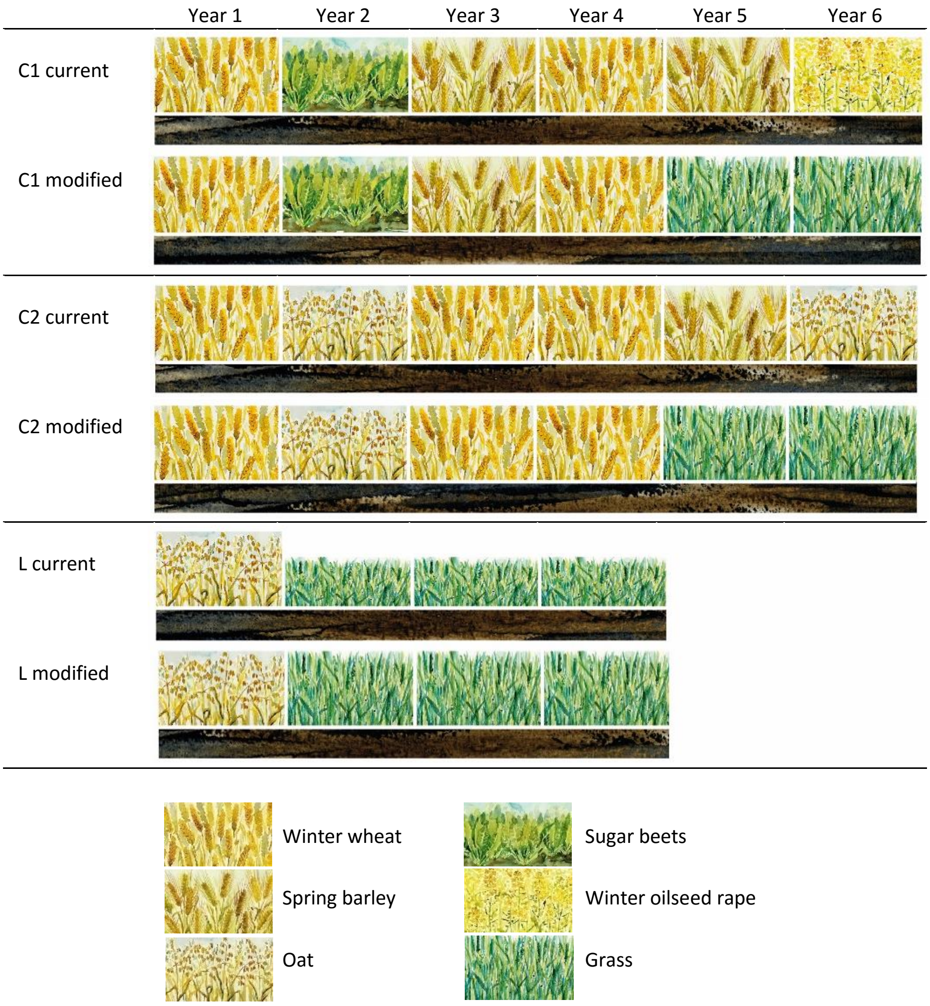

Figure 4. Schematic view of the crop rotations reflecting the current and modified crop production in the study regions. Illustration: Anna Persson (Ekologigruppen).

6.2 REGIONAL CROP ROTATION POTENTIAL

The present use of arable land (average for 2010-2014) in the regions evaluated (Table 41) are shown below (Olsson, 2015). The left graphs show the area of arable land in each region, and how much of that arable land that could be part of the chosen current crop rotation. For C1:c, the area under the current crop rotation is 197 000 hectares (ha, 1 ha = 10 000 m^2) out of totally 352 000 ha arable land, and for C2:c 77 000 ha out of totally 238 000 ha. The graphs on the right shows the impact of fully implementing the modified crop rotations on these 197 000 and 77 000 ha for C1:m (Figure 5) and C2:m (Figure 6).

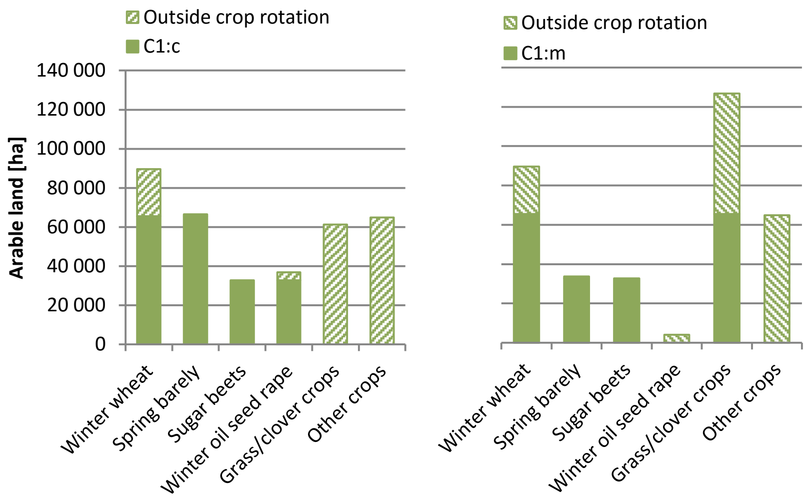

Figure 5. Current arable land use in region C1 (left), and after modifications including grass (right).

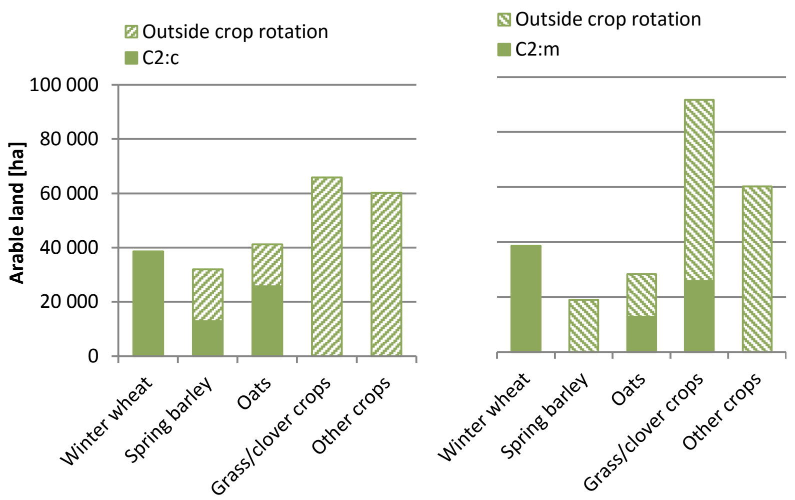

Figure 6. Current arable land use in region C2 (left), and after modifications including grass (right).

In the livestock region, grass/clover crops are currently occupying 94% of the arable land (average for 2010-2014), or 69 600 ha out of totally 74 000 ha (Figure 7), and the area used for grass cultivation was not assumed to change in the modified scenario.

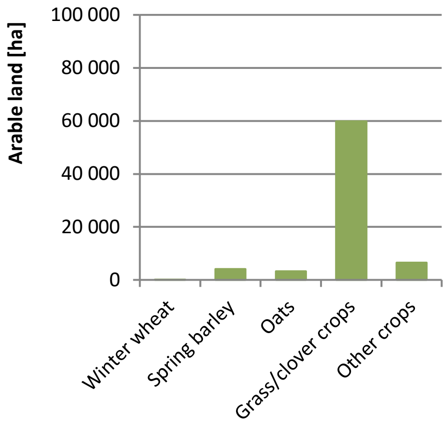

Figure 7. Current arable land use in region L

6.3 CROP PRODUCTION

6.3.1 Grass

Grass is assumed to be either cultivated as high-quality livestock feed or as biogas feedstock with relatively short growth periods (based on 2-3 harvests per year).

In the cereal regions C1 and C2, grass is grown solely as biogas feedstock in the modified crop rotations (C1:m and C2:m). It is undersown in the previous crop, i.e. winter wheat. The first production year is a full production year, where the grass is harvested three times, resulting in high biomass yields combined with high methane potentials. The second year is the break year, when the crop is ploughed up after the second harvest in order to allow for a autumn crop to be established.

In the livestock region L, grass is undersown in oats. In the current livestock scenario (L:c), the first two production years are full production years, where the grass is harvested three times, resulting in high biomass yields combined with high feed quality. In L:m, the two full production years are cultivated for high quality feed, as in L:c, while the third year grass is harvested as biogas feedstock. This partinioning was chosen for simplifying calculations. In practise, a different partioning would be chosen, e.g. the first two cuts as feed and the third a biogas substrate. In both scenarios, the third year is the break year, when the crop is ploughed up after the second harvest in order to allow for a autumn crop to be established.

For harvest, grass crops are cut and windrowed for field drying to 35% DM content. In the case of feed production, the crop is then coarsely chopped (ca 16 mm) by a forage harvester, while for the utilization as biogas feedstock, the grass is finely chopped (ca 4 mm). The grass biomass is collected by tractor-drawn field trailers during chopping. The feed biomass is transported to the farm, while the biogas substrate biomass is transported to the biogas plant. In both cases, the feedstock is loaded into a bunker silo and compacted. For ensiling, the silo is finally covered with a plastic sheet. From the bunker silo the ensiled biomass is then loaded for feeding of the cows and the biogas plants, respectively.

6.3.2 Other crops

The production of food and feed crops included field operations as recommended for the specific crops (Hansson *et al.*, 2014). For more details please refer to Appendix B.

6.4 CROP YIELDS

For grass crops no normal harvest level data for high-intensity production was available, since official statistics include even low and medium-intensity production systems, as well as organic production systems (SCB, 2014b). Therefore, expectable biomass yields have been estimated based on variety field experiments hosted by the Field Research Unit (FFE) at the Swedish University of Agricultural Sciences. For details please see Appendix B.

Table 3. Grass biomass yields [kg dry matter (DM) ha^{-1}] in the study regions as estimated from variety field experiments.

	Year 1				Year 2				Year 3		
Cut	**1**	**2**	**3**	**∑**	**1**	**2**	**3**	**∑**	**1**	**2**	**∑**
C1:m	4559	2826	2301	**9686**	4317	2000		**6317**			
C2:m	5288	2780	2829	**10897**	4526	2264		**6791**			
L:c	3352	1861	1944	**7157**	3151	1750	1691	**6592**	1953	2164	**4117**
L:m	4513	2505	2617	**9634**	4242	2356	2276	**8874**	2629	2913	**5543**

For the other crops, yields in the form of grains, seeds and beets were assessed based on normal yields as reported in official statistics (Appendix B). Depending on their position in the corresponding crop rotation, yields were adjusted for pre-crop effects (Table 4). For example, winter wheat yields in the first year of the crop rotation was affected by a pre-crop effect from winter oilseed rape (C1:c) and grass (C1:m). For more details please refer to Appendix B.

6.5 CROP FERTILIZATION

In the current cereal scenarios, all crops are assumed to be fertilized with mineral fertilizer. In the modified scenarios, mineral fertilizer is partly replaced by digestate from biogas production from grass. The calculations behind amounts and compositions of available biofertilizer are presented in Chapter 7.2. The biofertilizer was applied as the second application in winter wheat and grass (1 May and 1 June, respectively) in order to minimize crop damages due to soil compaction. The choice to empty the digestate storage before summer was also made to minimize the risk of methane leakage from storage.

In the current livestock scenario, manure from milk cows corresponding to 0.7 animal units per hectare (average livestock density for the region, see Figure 36) is applied to the grass crops as biofertilizer. In the modified scenario, the manure is instead used for biogas production together with the additionally produced grass biomass. The resulting digestate is applied as biofertilizer, partly replacing mineral fertilizers. The biofertilizers were applied as the second application in grass (1 June) in order to minimize crop damages due to soil compaction.

Amounts of nitrogen applied were calculated based on official recommendations at the expected biomass yields (SJV, 2014). Amounts of phosphorus and potassium were calculated based on typical biomass content (SJV, 2010). For more details please refer to Appendix B.

Table 4. Crops and final yields [kg ha^{-1}] in the studied crop rotations. Standard moisture content was assumed for cereals (14%) and oilseed rape. Sugar beets are reported as wet weight (22% DM content). Grass biomass yields are given as dry matter (DM) yields.

	C1:c	C1:m	C2:c	C2:m	L:c	L:m
Year 1	Winter wheat	Winter wheat	Winter wheat	Winter wheat	Oat	Oat
	7970	7039	5775	5426	3482	3482
Year 2	Sugar beets	Sugar beets	Oat	Oat	Grass, year I	Grass, year I
	58903	58903	4208	4208	7157	9634
Year 3	Spring barley	Spring barley	Winter wheat	Winter wheat	Grass, year II	Grass, year II
	5898	5898	5775	5891	6592	8874
Year 4	Winter wheat	Winter wheat	Winter wheat	Winter wheat	Grass, year III	Grass, year III
	6574	6574	4961	4961	4117	7742
Year 5	Spring barley	Grass, year I	Spring barley	Grass, year I		
	4968	9686	4581	10897		
Year 6	Winter oilseed rape	Grass, year II	Oat	Grass, year II		
	3876	6317	4208	6791		

7 Biogas production

7.1 GRASS PROPERTIES AS BIOGAS FEEDSTOCK

The grass properties at harvest are shown in Table 5. The properties are assumed to be the same for all harvest times in this study, based on that the growth periods are in the same range (42-56 days) due to frequent harvests. At wilting, ensiling and handling, biodegradation and losses will occur, changing the properties of the grass as biogas feedstock (Table 5). Further descriptions of underlying calculations can be found in Appendix B, where also the reasoning behind the selected grass properties and methane yields are described.

The theoretic methane yield shown in Table 5 is calculated based on the chemical composition, and this value is used as the maximum methane potential (B_0) in the calculation of methane emissions for the digestate according to the IPCC guidelines (IPCC, 2006). The biochemical methane potential (BMP) is based on data from experimental determinations in laboratory scale (see Appendix B), and this value is used as basis for calculating the full scale methane yield, which is assumed to be 90% of the BMP value.

Table 5. Grass composition before and after biochemical changes during wilting, ensiling and aerobic deterioration[2]

	Composition					Methane yield [1]	
	VS	C	N_{tot}	P	K	theoretic	BMP
	[% of DM]					[L (kg VS)$^{-1}$]	
Grass at harvest	91	45	3.1	0.35	1.8	363	334
Silage at feed-in	90	47	3.5	0.39	2.0	379	348

[1] Gas volumes are given as dry gas at 0°C and 101 kPa

[2] The DM of the silage after losses will be 33%, the total loss of mass is 3.3%, the DM loss 9.5% and the loss in methane potential 6.7%. In addition to the compounds shown above, the concentrations of the micronutrients Fe, Co, Mo and Ni are important for a well-functioning biogas process, and are shown in Table 26, Appendix B.

7.2 BIOGAS PRODUCTION IN CEREAL REGIONS

In the scenarios based on the modified crop rotations (C1:m, C2:m), the produced grass was used as biogas feedstock, and the calculations were based on a biogas plant using grass as single feedstock. In practice, biogas production from a single type of feedstock is rare, and a more likely scenario is that grass would be used as a co-substrate together with other types of waste or residues from agriculture, so called co-digestion. Calculations based on a completely grass based biogas plant, however, allows costs and environmental impacts to be calculated and attributed solely to the grass feedstock and back to the arable land in a transparent way.

For both C regions, two sizes of biogas plant were evaluated and compared. The size for the larger plant, 172 TJ a^{-1} (based on the lower heating value for methane, 35.8 MJ m^{-3}), is selected based on estimates of scale effects for biogas upgrading cost (Chapter 0). The other plant has half this size, 86 TJ a^{-1}, which is more commonly occurring today, where the average biogas production in the 35 existing co-digestion plants in Sweden in 2014 was 76 TJ a^{-1} (Chapter 0).

The set limits for the process design, and the actual operating conditions based on the process model calculations (Chapter 0) are summarized in Table 6.

Table 6. Process design parameters

Parameter	Unit	Set limit	Actual
OLR [a]	kg VS_{rem} m^{-3} d^{-1}	3	2.5
HRT	d	50	50
DM in digester [b]	%	9.5	7.8
TAN [c]	g l^{-1}	5.0	5.0
Biogas production [d]	m^3 h^{-1}	1 000	1 000

[a] defined as the mass of organic material removed in the process per reactor volume and day, indicated as VS_{rem}) (Lantz *et al.*, 2013)
[b] data from full-scale, crop-based biogas-plant monitoring presented by FNR (FNR, 2010) show that the viscosity of the reactor content increased significantly above 10% DM, causing problems in stirring.
[c] TAN stands for total ammoniacal nitrogen, so both NH_3 and NH_4^+
[d] The given value was used to determine dimensions of the larger (172 TJ a^{-1}) biogas plant. A plant half this size was also evaluated.

The outcomes in terms of digester volume, added feedstock and biogas production are summarized in Table 7. The demand of silage is the same for both regions since the grass properties are assumed to be the same for both regions. The demand of arable land is, however, different since yields are different, and transport distances differ due to differences in both share of arable land to total land area, and the share of arable land assumed to be used for the modified crop rotations.

Table 7. Process outcomes for the two plant sizes based on limits and actual outcomes presented in Table 6.

Plant size		86 TJ a-1	172 TJ a-1
Digester volume [a]	[m^3]	7 070	14 130
OLR	[kg DM m^{-3} d^{-1}]	3.9	3.9
	[kg VS m^{-3} d^{-1}]	3.5	3.5
HRT	[d]	50	50
Added to plant [b]	[t a^{-1}] grass silage	26 210	52 420
	[t a^{-1}] water	17 620	35 250
Biogas production	[m^3 h^{-1}]	500	1 000
Biogas to upgrading [c]	[m^3 h^{-1}]	478	955
Upgraded methane [d]	[m^3 h^{-1}]	262	525
	[TJ a^{-1}]	82.2	164
Arable land use C1 [e]	[ha]	3 550	7 101
Arable land use C2 [e]	[ha]	3 212	6 424
Transport distance C1 [f]	[km]	5.0	7.1
Transport distance C2 [f]	[km]	7.7	10.9

All gas volumes are given as dry gas at 0ºC and 101 kPa.
[a] The active volume is assumed to be 85% of this total digester volume.
[b] Grass silage with 32.7% DM. The amount of grass (35% DM) transported to the plant for ensiling is 27 110 and 54 220 t a^{-1}.respectively. For information on losses during ensiling and handling see Appendix B.
[c] After subtraction of leakage in biogas plant of 0.5% and 4% biogas to flair.
[d] After subtraction of 0.1% loss of methane in upgrading
[e] Given area is total areal of the 6-year crop rotations. Grass is produced two of these years, so on 1/3 of this area.
[f] One-way transport distance, see Appendix D for calculation.

The digestate amount and composition is shown in Table 8. Digestate is assumed to be stored under roof cover, and the amount and composition after storage losses (as applied to field) are also shown.

Table 8. Digestate amount and composition

	From digester	After storage losses
Amount [t a^{-1}]		
plant size 86 TJ a^{-1}	38 249	38 208 [a]
plant size 172 TJ a^{-1}	76 498	76 417 [a]
Composition		
DM [%]	7.8	7.7 [a]
VS [%]	5.5	5.4 [a]
C [% DM]	55.4	55.6 [a]
N-tot [kg t^{-1}]	7.8	7.7 [b]
TAN [kg t^{-1}]	5.0	4.9 [b]
P [kg t^{-1}]	0.9	0.9
K [kg t^{-1}]	4.5	4.5

[a] After loss of organic material due to production of CH_4 and CO_2 during storage (see Appendix D section 00)
[b] After a loss of NH_3-N corresponding to 1% of N-tot (Karlsson and Rodhe, 2002; SEPA, 2015) and no loss of N as N_2O (IPCC, 2006)(see Appendix D section 00)

The energy input and emissions are summarized in Table 9. For background information on the selected data see Appendix D.

Table 9. Energy input and emissions, biogas production in the cereal regions

Process		
Heat	119	MJ t^{-1} feedstock
Electricity	29	MJ t^{-1}
Methane leakage process	0.5%	of total methane production
Biogas to flair	4%	of total biogas production
Methane leakage flair	2%	of methane to flair
Upgrading		
Electricity	0.4	MJ m^{-3} biogas to upgrading
Heat	2.2	MJ m^{-3} biogas to upgrading
Methane slip	0.1%	of methane to upgrading
Compression		
Electricity	0.9	MJ m^{-3} upgraded gas
Distribution/tankstation		
Drivmedelsförbrukning	0.3 [a]	MJ m^{-3} upgraded gas
Electricity	0.25	MJ m^{-3} upgraded gas
Digestate		
Fuel for transport [b]	16	MJ km^{-1}
Fuel for loading	1.8	MJ t^{-1} digestate

[a] Based on an assumed return transport distance of 100 km
[b] For a vehicle with a loading capacity of 35 t, average for full transport with empty return.

7.3 BIOGAS PRODUCTION IN THE LIVESTOCK REGION

In the livestock region, the biogas scenario is based on an existing biogas plant where the current feedstocks are animal manure together with some additional waste fractions (Table 10). Waste that is suitable for biogas production is exposed to competition, so supply cannot be secured. In 2014, the biogas production at the plant was lower than the capacity. The assumption used in the present study is that in the modified scenario, the manure amount added remains the same as in 2014, but the waste currently used for biogas production is replaced altogether with grass. The digestate from manure and grass digestion is used as biofertilizer in the modified crop cultivation, where the current crop rotation is maintained, but the grass cultivation is intensified, with higher yields. This additional grass, from a modified crop rotation on an area of arable land corresponding to 3 000 ha (See Appendix E for details), corresponds to 14 000 t a^{-1} of silage after losses at handling/ensiling, Table 10.

Table 10. Feedstock for biogas production and the produced biofertilizers

	Biogas feedstock [t a^{-1}]			Biofertilizer [t a^{-1}]	
	Manure	*Waste*	Grass [a]	Manure [b]	Digestate [c]
		Waste is in scenario L:m replaced by grass			
Total weight	55 055	*12 451*	14 014	55 000	64 380
C	1 832	*631*	2 137	1 808	1 966
N-tot	222	*48*	159	215	379
NH4-N	106	*3*	0	98	254
P	39	*8*	18	39	57
K	221	*17*	92	221	312

[a] After subtraction of losses during handling and ensiling, se chapter 0

[b] Ingoing values for manure is from the full scale biogas plant[3] . When manure is assumed to be used as biofertilizer (without biogas production) in Scenation L:c, losses at storage are assumed to occur (sorage under floating crust), corresponding to loss of NH_3-N corresponding to 3% and of N_2O-N to 1% of N-tot, (Karlsson and Rodhe, 2002; IPCC, 2006) .

[c] After loss of organic material due to production of CH_4 and CO_2 during storage and loss of NH_3-N corresponding to 1% of N-tot (Karlsson and Rodhe, 2002; SEPA, 2015) and no loss of N as N_2O (IPCC, 2006)(see Appendix E, section 0). The ingoing B_o for cattle manure is based on reported methane yield in the full scale plant[3] (240 m3 (t VS)$^{-1}$, which is assumed to be 90% of B_0.

The manure and grass feedstock added to the existing biogas plant would give operating conditions as shown in Table 11. For comparison, the actual operating conditions for the plant in 2014 are given[3]. Both organic loading rate (OLR), ammonia-level (given as TAN) and DM in the process are higher than at current conditions, but are deemed possible for stable operation. The outcomes in scenario L:m are given divided after feedstock type to allow calculation of climate impact separately for the grass feedstock. The energy input and emissions are shown in Table 12 and are based on actual operational data from 2014 [3]. The same input data is used for the biogas production from waste/grass in scenario L:m.

[3] Björn Goffeng, Göteborg Energi. Personal communication October 2015.

Table 11. Process conditions in the existing biogas plant in 2014 (operating on manure and waste) and with grass replacing waste as in scenario L:m. The outcomes in the modified scenario is divided based on the feedstock, and the contribution from manure is the same as in the current scenario.

		Actual conditions 2014	Scenario L:m (manure and grass)	
Total digester volume [a]	[m^3]	9 400	9 400	
OLR	[kg DM m^{-3} d^{-1}]	1.9	3.0	
	[kg VS m^{-3} d^{-1}]	1.6	2.6	
HRT	[d]	43	42	
TAN	[kg t^{-1}]	3.0	4.0	
DM	[%]	4.2	6.3	
Divided after feedstock		manure +waste	Manure	grass
Feedstock addition	[t a^{-1}]	67 506	55 055	14 014 [b]
Feedstock DM	[%]	8.2	7.6	33
Biogas production	[m^3 h^{-1}]	276	157	267
Biogas to upgrading [c]	[m^3 h^{-1}]	231	132	224
Methane content	[%]	62	60	55
Upgraded methane [d]	[m^3 h^{-1}]	144	79	123
	[TJ a^{-1}]	45.1	24.8	38.5

All gas volumes are given as dry gas at 0°C and 101 kPa.
[a] The 400 m^3 covered post-digester with gas collection is included in the total reactor volume. The active volume is assumed to be 85% of the total digester volume.
[b] After subtraction of losses during ensiling and handling. The wilted grass transported to the biogas plant amounts to 9 642 t with 35% DM.
[c] After subtraction of leakage in biogas plant (0.32%) and biogas to flair (4%).
[d] After subtraction of 0.18% methane slip in upgrading.

Table 12. Energy input and emissions, biogas production in the livestock region

Process		
Heat	139	MJ t^{-1} feedstock
Electricity	42	MJ t^{-1}
Methane leakage process	0.3%	of total methane production
Biogas to flair	4%	of total biogas production
Methane leakage flair	2%	of methane to flair
Upgrading		
Electricity	3.1%	of energy in upgraded biogas
Heat	1.30	MJ/m3 biogas to upgrading
Methane slip	0.2%	of methane to upgrading
Compression		
Electricity	4.2%	of energy in upgraded gas
Distribution		
Fuel	0.6	MJ m^{-3} upgraded gas
Electricity	0.25	MJ m^{-3} upgraded gas

[a] based on a return transport distance of 200 km

8 Economic assessments

8.1 CROP PRODUCTION

Economic assessment of the current and modified crop rotations was carried out for each of the study regions. The assessment used total step calculations in order to calculate total costs from machinery use, use of buildings and utilization of production means (Appendix G).

Costs for grass as biogas substrate include cultivation, harvest, transport to the biogas plant, storage and feed-in into the digester. Economic costs for digestate transport from the biogas plant to a storage at the farm was included in the costs for the biogas production. However, costs for the storage and spreading of the digestate were included in the crop production costs.

Cost included cost for fertilizers, liming agents, seeds, pesticides, and machinery use (including capital costs, fuel and salary costs for drivers). Costs originating from use of buildings was included by investment calculations at an interest of 6%, breaking down the costs per cubic meter storage capacity used. Costs of other buildings, general work on the farm, farm subsidies and tenancy were excluded.

Revenues from crop sales of food and feed crops were calculated using prices of 2014 according to the Swedish price index (SCB, 2015), Table 13. Revenue from grass biomass was calculated differently between regions. In the cereal regions C1 and C2, the required sales price for grass biomass was set to result in an equal economic result for the whole crop rotation compared to the current crop rotation. This unchanged economic outcome was assumed necessary in order for farmers to adopt grass production.

Table 13. Crop prices according to the Swedish price index (SCB, 2015).

Crop	Assumed sales price [€/t][a]
Spring barley	148
Spring oat	121
Sugar beets	26
Winter oilseed rape	294
Winter wheat	156

[a] Price at standard moisture content (cereals, 14%, oilcrops 9%, grass 0%, sugarbeets 78%).

In the livestock region L, the situation was different, since grass was already part of the crop production in the current scenario. Grass produced in L:c was assumed to be used as fodder on the farm. In L:c, the internal price was set so that it was that same as the costs of grass biomass production, i.e. no profit was made in this step of livestock production. On the other hand, the price of grass biomass in the modified scenario L:m was calculated in two ways:

(1) The sales price was set to produce the same economic result as in the L:c scenario: (a) as a common price for fodder and biogas substrate production, where the economic profit of intensification was applied to both feed and biogas substrate; and (b) as a subsidized price where the price of fodder production was constant and all profit from intensification was applied to the biogas substrate.

(2) The price remained the same as in the L:c scenario for years 1 and 2 (feed production, no profit) and was set to 100 €/t for year 3 (biogas substrate).

8.2 BIOGAS PRODUCTION

In general, the economic feasibility for biogas systems depends on the production cost versus the possible income from the biogas and the digestate produced. These two parameters are affected by site specific conditions and the assumptions made in this study are presented below.

Biogas production cost depends on the investment in the biogas plant and associated capital cost, operation and maintenance, process energy and the cost for feedstock. The latter is of particular importance for biogas systems based upon energy crops such as grass crops. Also, feedstock properties and estimated biogas production rate affect the overall production cost.

8.2.1 Investment and capital cost

The investment cost for the biogas plant depends for example on process design, reactor configuration and scale of operation. In this study, the investment cost for a biogas plant using energy crops only is calculated based on the findings presented in Lantz et al. (2013), see Figure 8.

Given the calculated active reactor volume of approximately 6 000 and 12 000 m^3 in C1 and C2, as presented in chapter 7.2, the corresponding investment cost is calculated to 2.0 and 3.6 million € respectively. Based on a discussion with a biogas plant supplier in Sweden[4], this calculated cost seems to be in the same order as actual plants built in Sweden although different choices in the design process affect the overall investment.

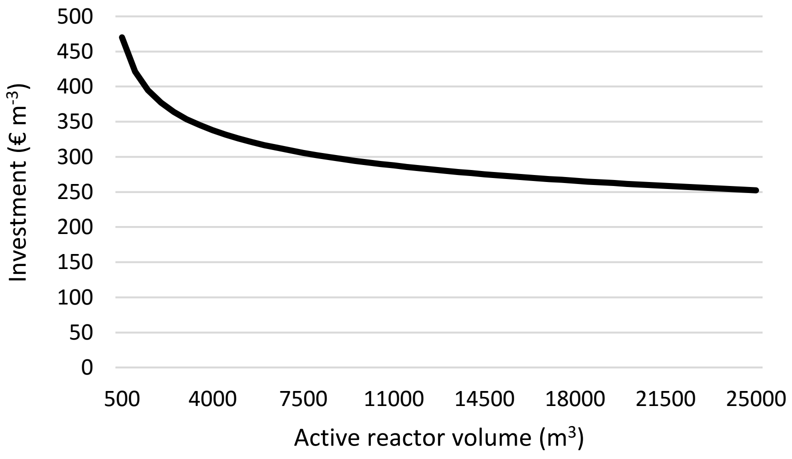

Figure 8. Calculated investment in biogas plant (Lantz, 2013).

Regarding the upgrading plant, the investment cost depends on chosen technology as well as scale. Based on contacts with industrial actors in Sweden, the cost for a chemical

[4] Olof Pettersson, Purac AB. Personal communication.

scrubber with a capacity of 500 and 1000 $m^3 h^{-1}$ is set to 1.8 million and 2.6 million € respectively [5].

The capital cost for the biogas plant as well as the upgrading plant are calculated based on an annual interest of 6 % and 15 years' depreciation for both investments.

In the L:m scenario it is assumed that the biogas plant is equipped with a separate feed in, bypassing the hygienization tanks. The investment cost is estimated to 0.2 million € as presented in Appendix E.

8.2.2 Process energy

In this study, the cost for electricity and heat are set to 69 €/MWh and 15 €/GJ respectively. For additional background information see Appendix F.

8.2.3 Operation and maintenance

Based on the literature review presented in Lantz et al. (2013) the cost for operation and maintenance of the biogas plant was set to 5 €/t feedstock corresponding to approximately 5 % of the investment. For upgrading the annual cost was set to 0.3 and 0.2 €/GJ respectively [6]. For the compressor station the cost was set to 3 % of the investment (Lantz, 2013).

8.2.4 Transportation of digestate

In addition to biogas, the biogas plant also produces liquid digestate which is assumed to be transported by truck to the farmers delivering grass crops and utilized as fertilizer.

In this study it was assumed that the digestate is transported by truck with a loading capacity of 35 t. Average speed was set to 50 km/h and digestate loading/unloading was assumed to take 0.25 h/35 t. Transportation cost was calculated using 100 €/h for transport as well as for loading and unloading.

In C1 and C2 the one-way transportation distance varied between 5 and 11 km resulting in a transportation cost between 1.1 and 1.6 €/t.

In the L:m scenario, the transportation distance is calculated to 11,4 km with a corresponding transportation cost of 1,6 €/t.

8.2.5 Income from biogas

In this study it was assumed that the biogas producer could sell upgraded and compressed biogas at the biogas plant for 21 €/GJ. The effect of a higher or lower gas price are also evaluated in the sensitivity analysis. Since this number is not presented publically it is estimated based on public available market prices and distribution cost reported in the literature. For more information, see Appendix F.

8.2.6 Income from digestate

The digestate produced is utilized as fertilizer replacing mineral fertilizers. As such, is has an economic value. In this study, however, it is assumed that the farmers receive

[5] Lars-Evert Karlsson, Purac Puregas. Personal communication.
[6] Lars-Evert Karlsson, Purac Puregas. Personal communication.

the digestate for free which decrease the farmers' expenses and thus decrease the production cost for crops. This simplified assumption may, however, not be applicable in reality since other farmers may be able to pay a higher price for the digestate given their individual conditions.

8.3 SOCIOECONOMIC EVALUATION

The proposed change from the current to the modified scenarios will impact greenhouse gas emissions as well as other environmental aspects. The impact of the proposed modifications is evaluated by assigning the total environmental impact a socioeconomic cost. This cost is presented as a range based on information from previous studies, except for the CO_2-emissions, which are valued according to the current value of the CO_2-tax. The used values are shown in Table 14, and the background information can be found in Appendix H.

Table 14. Socioeconomic values used in this study

Environmental impact	Indicator	Socioeconomic cost	Source
Global warming potential	kg CO_2-eq	0.12 € (kg CO_2-eq)$^{-1}$	(Skatteverket, 2016)
Eutrophication potential	kg N-eq	0.4-7.8 € (kg N)$^{-1}$	(SEPA, 2009)
Acidification potential	kg SO_2-eq	3.1-4.9 € (kg SO_2-eq)$^{-1}$	(Trafikverket, 2016)
Particle emission	kg particles (PM2.5)	62-484 € kg^{-1}	(Trafikverket, 2016)

9 Results & discussion

9.1 SOIL ORGANIC CARBON CHANGES

The soil carbon development was found to differ considerably between the studied regions (Figure 9). The average initial SOC content was lowest in the C1 region (1.6 %), followed by the C2 region (2.0 %) and highest in the livestock region L (2.6 %). SOC content in the current crop rotations was roughly at steady state for C1, decreasing for C2 and increasing in L. When crop rotations where modified, SOC content increased, and the effect of the crop rotation changes accounted for roughly 50% of the effect in C1 and C2. The other 50% of the effect were attributed to the application of carbon in the form of digestate. In C1, the transition from current to modified crop rotation led to an absolute increase in SOC. In C2, the transition resulted in a stabilization of the SOC. However, without digestate application, the SOC content in C2 would have been decreasing.

In the livestock region L, differences between the current and the modified crop rotation were marginal. Digestion of the manure would likely lead to a decrease in carbon applied to the soil, due to carbon removed as methane and carbon dioxide in the biogas process. However, in the modified scenario, the digestate also contains additional carbon from the grass biomass, which balances the removal of carbon. The amount of aboveground and belowground crop residues contributing to SOC remains unchanged as it consists of the same stubble after harvest and the same root biomass as limited by the SOC model assumptions.

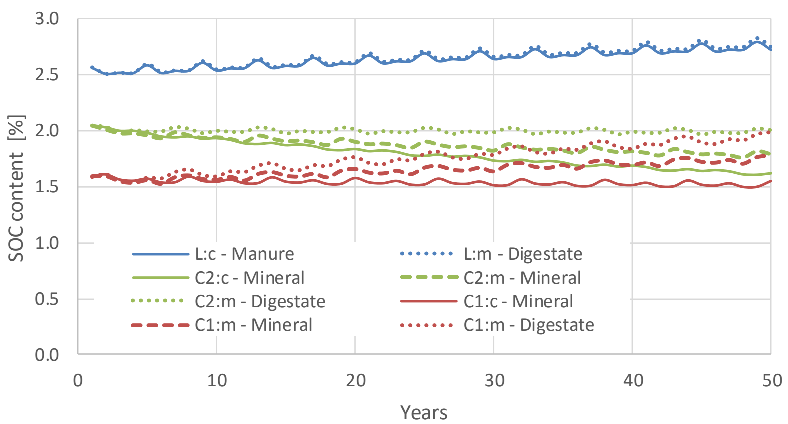

Figure 9. Development of soil organic carbon (SOC) content in the soils of the study regions under current and modified crop rotations.

The annual SOC effect, which is further used for the calculation of GHG and nitrogen emissions reflects this development (Figure 10).

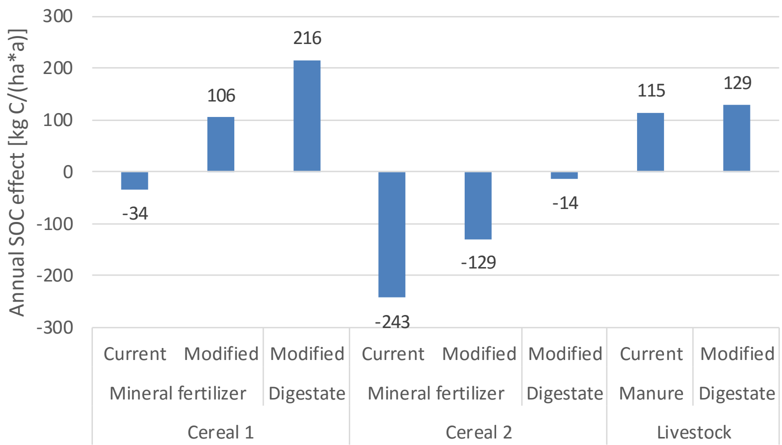

Figure 10. Annual soil organic carbon (SOC) effect in the soils of the study regions under current and modified crop rotations.

9.2 LAND USE

Implementing the modified scenario in region C1 on the arable land cultivated under the current crop rotation would mean using 65 600 ha of arable land for grass production. This would give a biogas production corresponding to 4.8 PJ a^{-1}. For the C2 region, the corresponding values are 25 800 ha and 2.1 PJ a^{-1}. In total, this would result in a biogas production of 6.8 PJ a^{-1}, which is more than the current production in all Swedish biogas plants (5.6 PJ in 2014, see Figure 42).

The crops replaced in the crop rotations would amount to 266 000 t DM a^{-1} cereals (whereof 2/3 spring barley) and 116 000 t DM a^{-1} oil seed rape seed. This can be compared to the production statistics for the last three years, where total cereal production has been 5.0-5.8 million t a^{-1}, whereof 2.2-2.5 million t has been used as animal feed and 1.0-1.9 million t has been exported (SJV, 2016a). The use of rape seed based biodiesel (RME) in Sweden 2014 was 15 PJ (Figure 11), which corresponds to 1 110 000 t DM rape seed (SEA, 2015a) [7]. Out of this, 7% was based on oil seed rape produced in Sweden. The conflict between the use of arable land for food/feed crops and energy is important to consider in any scenario involving the use of land or biomass, and a high share of the Swedish utilization of biofuels is today based on starch-rich (ethanol) or oil-rich (biodiesel) crops, mainly produced within the EU ((SEA, 2015a, b, c), Figure 11), while in other parts of Europe, oil-rich crops dominate (Marelli *et al.*, 2015).

[7] Assuming a yield of 0.32 t RME per t rape seed with 91% DM

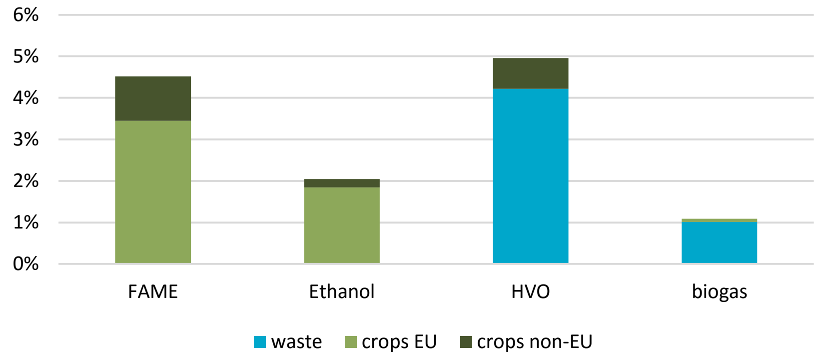

Figure 11. Biofuels used in Sweden 2014 as share of the total energy used in domestic transport. Fuels produced from crops from arable land (within and outside EU) are shown in green, waste based fuels are shown in blue.

At the same time, the arable land in the cereal regions under current conditions loose soil organic matter, especially pronounced in the C2 region (Figure 10). Mineral agricultural soils in Sweden on average loose SOC corresponding to 60 kg CO_2-eq (ha a)$^{-1}$ (SEPA, 2015). This in itself is not sustainable in long-term and measures against further SOC losses need to be taken at some point. While SOC losses may not necessarily directly lead to decreasing crop yields, these losses render arable land to be sources of carbon emissions. Therefore, the benefits from a climate perspective of converting the crop rotations as a whole to carbon sinks should also be taken into consideration when arable land use is discussed.

9.3 GREENHOUSE GAS EMISSIONS

9.3.1 Cereal regions

The greenhouse gas emissions for the current and modified systems in the cereal regions were calculated based on the emission data presented in Appendix I, and are shown in Figure 12. The emissions are given as average per ha for the crop rotation as a whole.

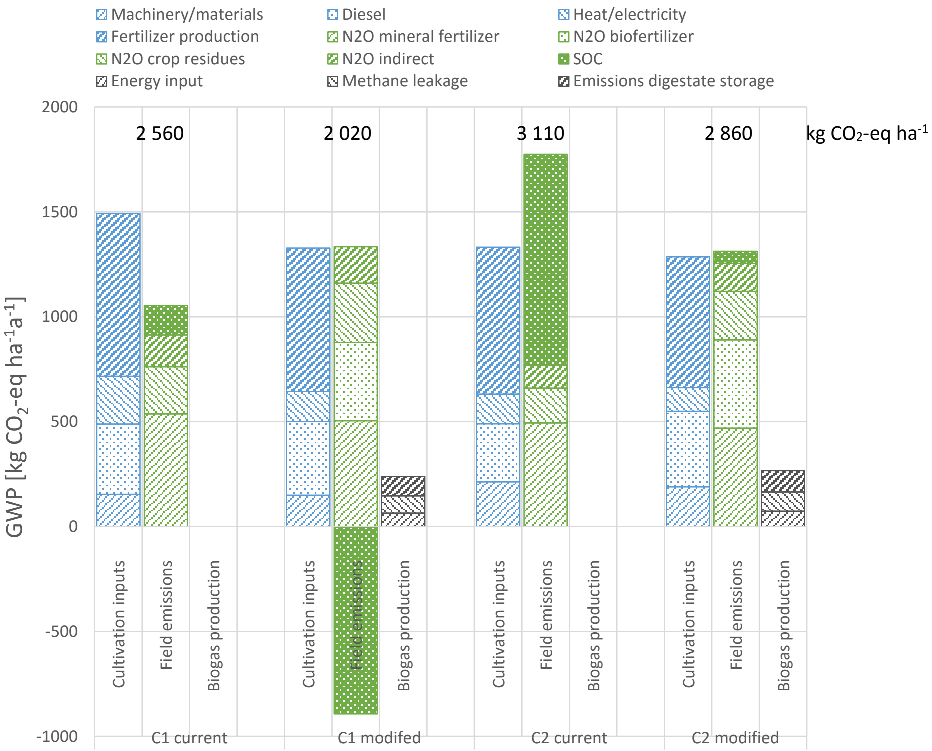

Figure 12. Greenhouse gas emissions for crop production and biogas production before systems expansions in C regions. Net values are given above bars. The data shown is for the smaller biogas plant (86 PJ a^{-1}). The difference for the scenario with the larger biogas plant was a minor increase in diesel consumption (+0.4% for C1 and +0.7% for C2) due to the longer transport distances for the grass, which does not impact the given GWP figures.

While emissions related to cultivation inputs decrease, the field N_2O emissions increase with 420 (C1) and 485 (C2) kg CO_2-eq ha^{-1} for the modified systems, both due to the biofertilization and because grass contributes with nitrogen containing crop residues. This emission increase is, however, balanced by the annual SOC effect, which decreases the emissions by 1034 (C1) and 947 (C2) kg CO_2-eq ha^{-1}. The positive impact of the increased addition of biomass (as biofertilizer and crop residues) thus largely outweighs the increase in N_2O emissions. The total impact is a reduced emission of 0.78 (C1) and 0.51 t CO_2-eq ha^{-1} for cultivation inputs and field emissions (these net emissions are shown as "cultivation" in Figure 13) when the modified crop rotations are introduced, given as average per ha and year in the whole crop rotation.

The emissions related to the production of biogas are small in comparison, 238 (C1) and 265 (C2) kg CO_2-eq ha^{-1}. In the systems expansion, emissions for cultivation of the lost crops elsewhere were added, and the benefits of applying the biogas for replacing diesel in heavy vehicles or busses were included (Figure 13). The total impact (the benefit of replacing fossil fuels minus the emissions in biogas production and cultivation of lost crops shown in Figure 13) is totally a reduced emission of 0.75 (C1) and 1.05 (C2) t CO_2-eq ha^{-1}. Also this value is given as average per ha and year for the whole crop rotation.

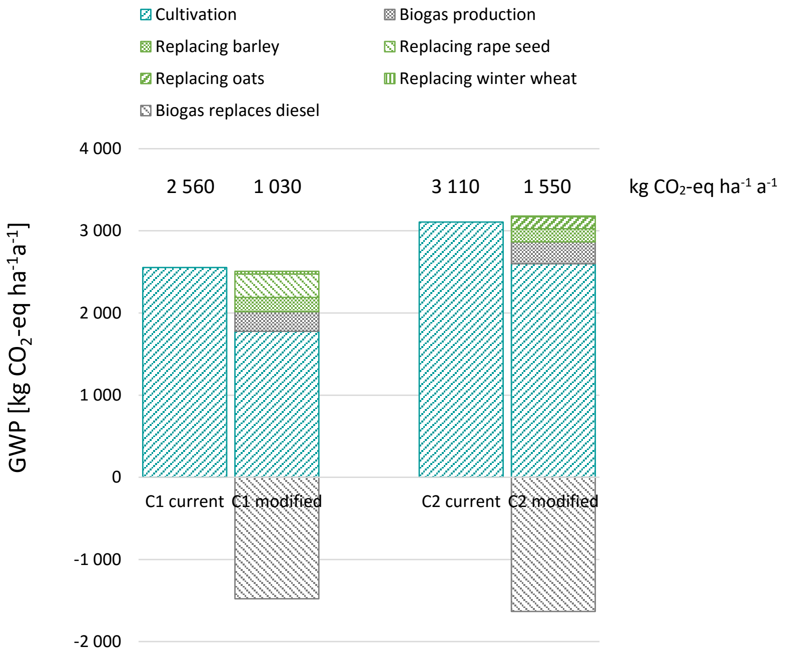

Figure 13. Greenhouse gas emissions after systems expansion. Net values are given above bars.

The total climate benefits for both regions are similar; avoided emissions of 1.5 t CO_2-eq ha^{-1} (Figure 13) of which roughly half can be attributed to the increased input of organic matter in cultivation, and half to the benefits of the renewable fuel for C1. For C2, 1/3 of the benefit can be attributed to cultivation impacts and 2/3 to the fuel.

Applying the modified scenarios on the total area of arable land under the current crop rotation in the two regions would give a climate benefit attributed to cultivation of 0.2 Mt CO_2-eq a^{-1} and to the renewable fuel of 0.2 Mt CO_2-eq a^{-1}. This can be compared to the total greenhouse gas emission from agriculture of 10 Mt CO_2-eq in 2013, whereof 2 Mt CO_2-eq was from the degradation of soil organic matter and release of SOC (SEPA, 2015). The total greenhouse gas emission in Swedish road transport in 2014 was 16 Mt CO_2-eq (SEA, 2015c).

For comparison, the greenhouse gas emissions were calculated according to the criteria in the EU renewable energy directive (RED) (EU, 2009). In EU RED, soil carbon impacts of the direct change of arable land use is not taken into account, so differences compared to the current land use are not considered. However, the EU RED methodology opens up for inclusion of SOC accumulation via improved agricultural management, such as increased use of organic fertilizers (EU, 2009), and the impact of the use of biofertilizer (manure and digestate) on SOC content was included. The result is shown in Figure 14.

For comparison, the results from Figure 13 were recalculated per MJ fuel produced. With ISO-methodology, the result shown for cultivation is the difference between the current and the modified scenarios, and the emissions from biogas production are shown separately, together with emissions from replacing lost crops through systems expansion (Figure 14). The latter (21 g CO_2-eq MJ^{-1} for C1, 12 g CO_2-eq MJ^{-1} for C2) would not be included in a calculation according to EU RED, since the impact of the loss of food/feed crops is not calculated in relation to a reference system based on the current land use. Instead, the competition for arable land would have been addressed by adding emission penalties, iLUC factors, if the crop used for biofuel production belonged to the group starch rich crops (12 g CO_2-eq MJ^{-1}), oil crops (55 g CO_2-eq MJ^{-1}) or sugar crops (13 g CO_2-eq MJ^{-1}). No iLUC factors have so far been discussed for crops other than these.

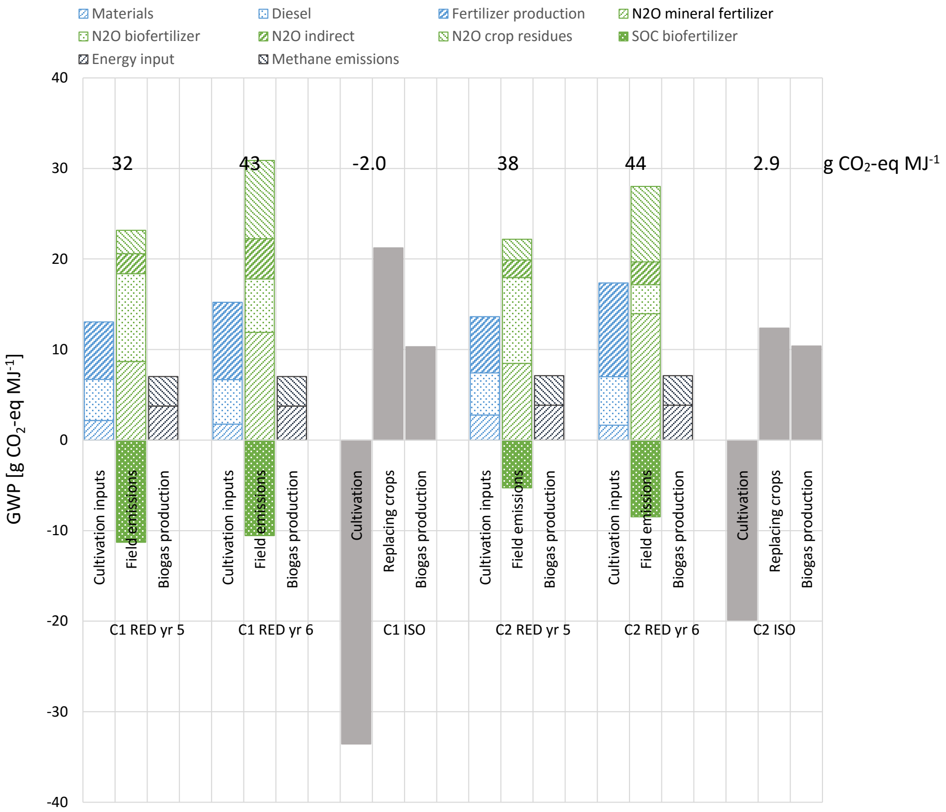

Figure 14. Greenhouse gas emissions per MJ produced fuel. Net values are given above bars. Results according to EU RED compared to ISO-methodology. In addition to the differences in methodology, different input values are used in the EU RED calculations for some of the emissions (higher for electricity and diesel use, lower for mineral fertilizer production), and somewhat lower GWP characterization factors (see Appendix H & I for further details).

Due to the large SOC impact in the ISO-based calculation, the net emission for the produced biogas was close to zero (-2 to 3 g CO_2-eq MJ^{-1}) for both regions. When EU RED methodology was used, excluding direct land use change impacts on SOC, the emissions per MJ fuel produced were on average 36 and 40 g CO_2-eq MJ^{-1} for C1 and C2, respectively. The demand on new biogas plants is an emission reduction of 60 % compared to the reference value 83.8 g CO_2-eq MJ^{-1}, corresponding to a maximum emission of 33.5 g CO_2-eq MJ^{-1}. The scenarios as designed in this study will exceed this limit. However, by e.g. replacing fossil fuel utilization in cultivation (representing 5 g CO_2-eq MJ^{-1}) and decreasing methane emissions in biogas production (representing 3 g CO_2-eq MJ^{-1}) the 2.6 (C1) to 6.6 (C2) g CO_2-eq MJ^{-1} decrease needed could be achieved. It should also be pointed out that the break year for the grass (yr 6) results in much higher emissions compared to a full production year of grass crops. This is mainly due to the high input of nitrogen containing crop residues to the soil. When mineralizing, these crop residues will contribute to N_2O emissions, but the benefits of the contribution to SOC are not to be included according to present methodology. Biogas as vehicle fuel from grass produced in a full production year will therefore easier fulfil the EU RED criteria (as is the case for region C1, yr 5, Figure 14). A prolonged cultivation of grass (3-6 years) may decrease impact of the break year. However, grass yields tend to decrease with each year of continued cultivation and the annual SOC effect would likely decrease as well, potentially outweighing the effect of prolonged grass cultivation.

In a sensitivity analysis, the impact of changing the GWP characterization factors to updated factors with a shorter time perspective was evaluated (Table 53)(IPCC, 2013). The base case value given in Figure 15 is the total value of the avoided emissions per ha in the crop rotation when changing from the current to the modified scenario. Changing the GWP for N_2O will impact both the current and modified scenario, and gives little net impact. The increased GWP with a 20 year time perspective for CH_4, however, decreases the climate benefit, and shows the importance of minimizing methane emissions. The results indicated as CH4- (Figure 15) are based on reducing the methane leakage in production from 0.5 to 0.1%, and the MCF from 3.5% to 0.5%. The MCF indicates the average share of methane that will be emitted from digestate storage in relation to the potential, and since the scenarios here are built on emptying digestate storages in early summer, applying a lower MCF is motivated. As shown by Rodhe *et al.* (2013), MCF is very temperature sensitive, and was in winter storage of digestate as low as 0.1%. On the other hand, annual average temperatures are increasing and could in Southern Sweden give a risk for higher temperatures also in winter. The results shown as CH4+ gives the impact of applying the IPCC default values for manure storage, with MCF of 10% (IPCC, 2006).

All in all, the change from the current to the modified scenarios will give a good climate impact, but awareness of the climate impact of methane is important, and irrespective of which time perspective that is considered for the GWP, methane emissions should be minimized to maximize climate benefits.

Figure 15. Sensitivity analysis of the total impact of changing from the current to the modified scenarios for C1 and C2.

9.3.2 Livestock region

For the livestock region, calculations have been made according to EU RED methodology only. The results are shown in Figure 16. For an existing biogas plant as the one used for the calculations in this case, the demand on reduction of greenhouse gas emissions is presently 35%. From 2018, this will be increased to 50%, or a maximum emission of 41.9 g CO_2-eq MJ^{-1}. The average emissions for all three production years was here 38 g CO_2-eq MJ^{-1}. Hence, the grass can be used as co-feedstock with manure in the existing biogas plant and fulfils the EU RED criteria on greenhouse gas emissions. To fulfil the demand on 60% reduction for new installations, the emissions need to be reduced by 4 g CO_2-eq MJ^{-1}, which for example could be achieved by replacing fossil fuel in cultivation (contributing with 5-6 g CO_2-eq MJ^{-1}) or reducing the emissions from energy use in production (contributing with 8 g CO_2-eq MJ^{-1}).

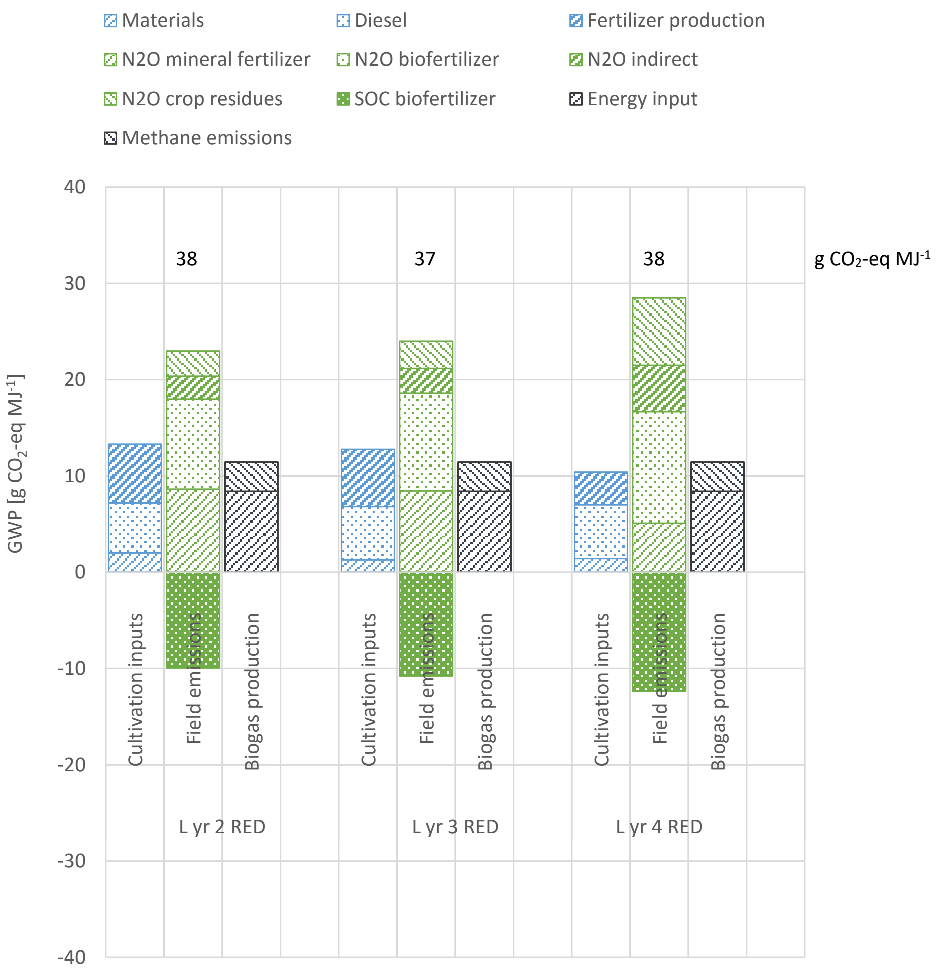

Figure 16. Greenhouse gas emissions for grass as biogas feedstock in the livestock region. The individual value for each of the three years in the crop rotation is given. Net value is given above the bars.

9.4 EUTROPHICATION, ACIDIFICATION AND PARTICLE EMISSIONS

The results for eutrophication potential, acidification potential and particle emissions are shown first for the crop and biogas production (Figure 17, 19, 21), and then in a second step including lost or gained products through systems expansion (Figure 18, 20, 22).

9.4.1 Eutrophication

Even though the leakage of nitrogen (as NO_3^-) to water decreased for the modified crop rotations when grass was introduced, the total emission increased due to the increased evaporation of ammonia when crop production was partly biofertilized, Figure 17. A change from the current to the modified scenario would increase the emissions of eutrophying compounds with 1.9 and 1.2 kg PO_4^{3-}-eq ha^{-1} a^{-1} for C1 and C2, respectively, Figure 18.

Variability in emissions depending on which application technique is used for biofertilizer, for which crop and the time of application in relation to the crop development, is large and needs to be addressed by suggesting viable application patterns. The ammonia emissions from roof covered digestate storages are assumed to be low, but emissions due to evaporation after field application of digestate in grass was assumed to be 20% of the added TAN. Reducing this emission to 10% would reduce the impact of the change from an emissions of 2.1 kg PO_4^{3-}-eq ha^{-1} a^{-1} to 0.9 (C1) and 1.4 kg PO_4^{3-}-eq ha^{-1} a^{-1} to 0.4 (C2), but would still give a net increase in emission.

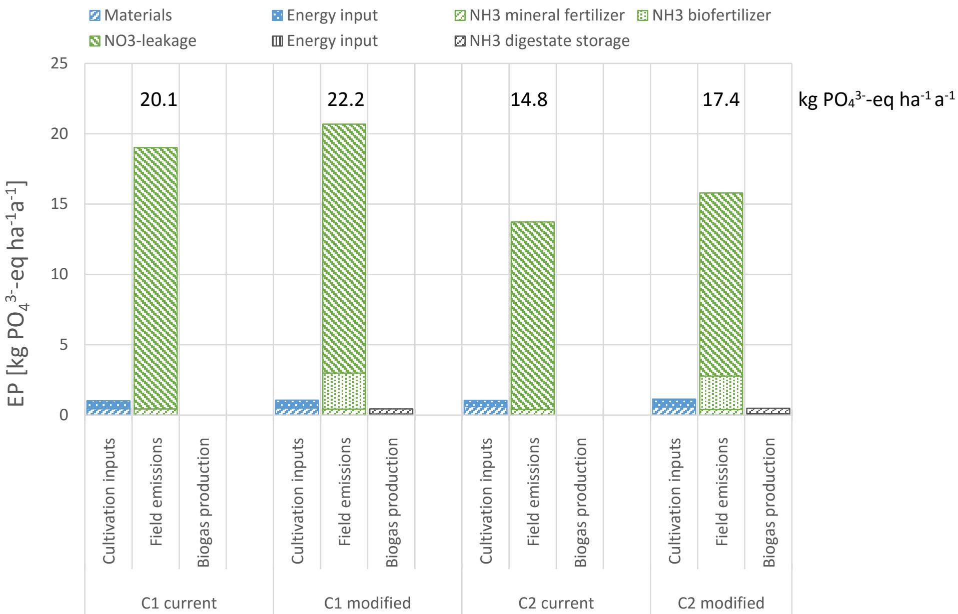

Figure 17. Eutrophication potential for crop production and biogas production before systems expansions in C regions. Net values are given above bars.

Figure 18. Eutrophication potential after systems expansion. Net values are given above bars.

9.4.2 Acidification

The increased evaporation of ammonia when crop production is partly biofertilized also largely impacts the acidification potential for the modified scenarios (Figure 19 and Figure 20), which only to a small degree is offset by the lower emissions of acidifying compounds when biofuel replaces diesel. A change from the current to the modified scenario will increase the emissions of acidifying compounds with 10.4 and 9.2 kg SO_2-eq ha^{-1} for C1 and C2 respectively. The same assumption for decreased ammonia evaporation after field application in grass as is described for eutrophication (decrease from 20% to 10%) would reduce these emissions to 4.8 (C1) and 4.5 (C2) kg SO_2-eq ha^{-1} a^{-1}.

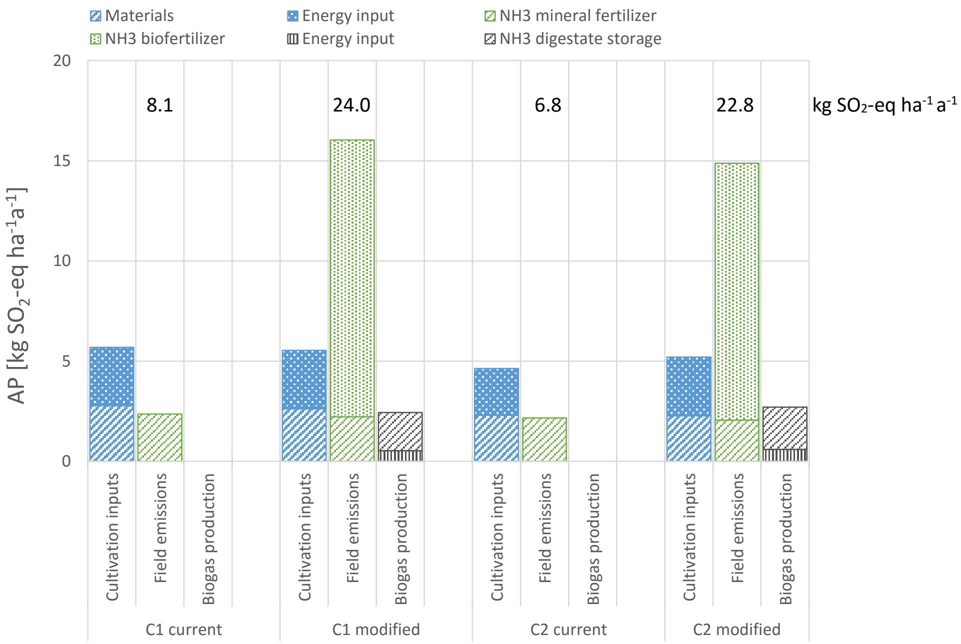

Figure 19. Acidification potential for crop production and biogas production before systems expansions in C regions. Net values are given above bars.

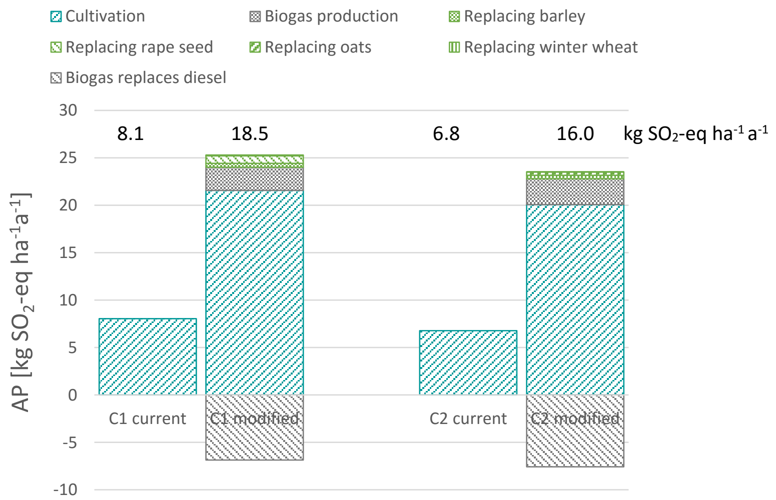

Figure 20. Acidification potential after systems expansion. Net values are given above bars.

9.4.3 Particles

The total particle emissions will decrease when the modified scenarios replace the current scenarios. Emissions from production of crops, biogas and materials like mineral fertilizers will increase in the modified scenarios when replacement of the lost crops are included in the systems expansion (Figure 21 and Figure 22). Replacing diesel with biofuel will, however, result in a decrease of particle emissions. This latter emission is handled separately in the socioeconomic evaluation.

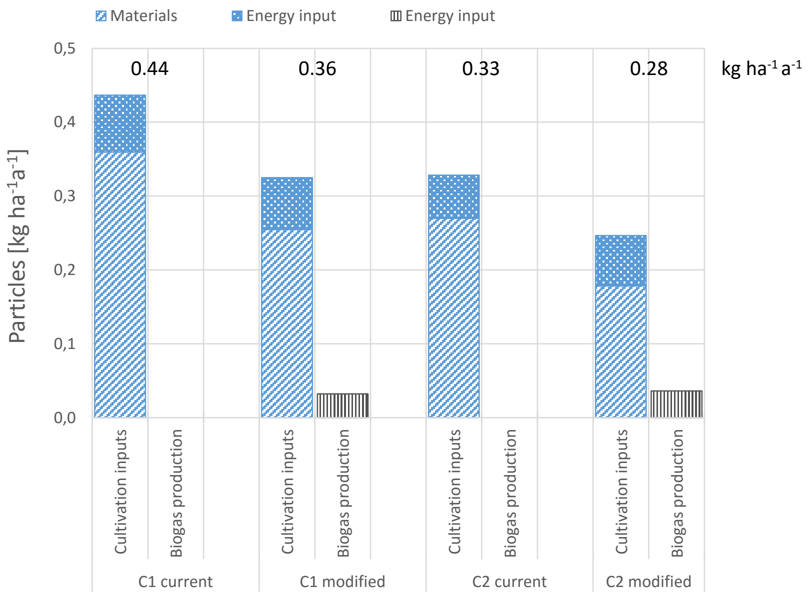

Figure 21. Particle emission for crop production and biogas production before systems expansions in C regions. Net values are given above bars.

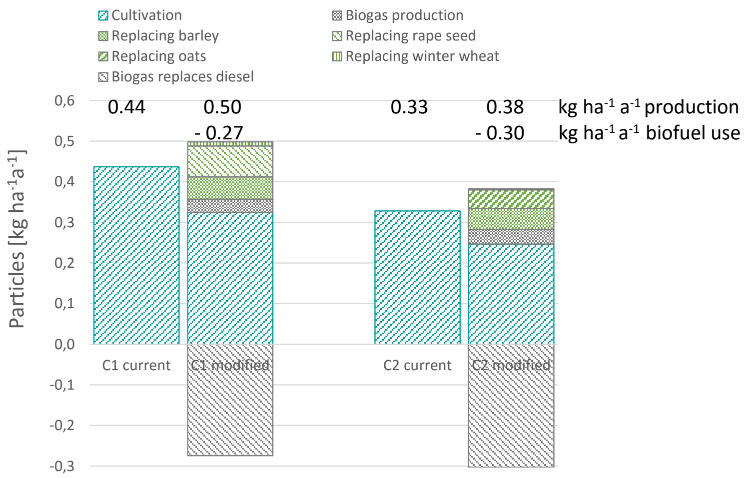

Figure 22. Particle emission after systems expansion. Net values are given above bars and are separated in emissions in production and in the use of biofuels.

9.5 ECONOMIC EVALUATIONS

9.5.1 Crop production

Production costs and revenue structures differed between current and modified crop rotations, but more significantly between study regions (Figure 23).

Costs were dominated by cultivation costs, while costs for harvest, transport and storage accounted for roughly one third of the total costs. Of the production means, machinery costs dominated total costs. Machinery costs increased in the cereal regions when changing from current to modified crop rotations, due to transport and handling of the digestate. At the same time fertilizer costs decreased. Fertilizer costs in the livestock region remained on roughly the same level, since the current scenario already included use of organic fertilizer, i.e. cattle manure.

By definition, the difference between costs and revenue in the cereal scenarios C1 and C2 remained constant. In the livestock region, revenues increased substantially, when revenues for L:m were based on the same revenue as in L:c for years 1 and 2 (144 €/t) and were set to 100 €/t for year 3 (biogas feedstock production), Figure 23.

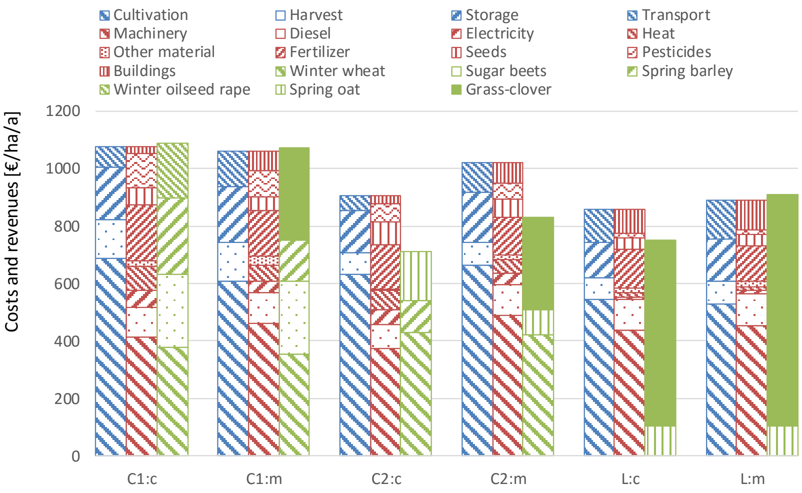

Figure 23. Production costs (partitioned for production stages (blue) and for production means (red)) and revenues (partitioned for individual crops (green)) for the current and modified crop rotation in the different study regions for the smaller biogas plant (24 GWh) in C1 and C2. Results for the larger biogas plant (48 GWh) in C1 and C2 differed only marginally. Revenues for L:m were based on the same revenue as in L:c for years 1 and 2 and were set to 100 €/t for year 3 (biogas substrate production).

These results were reflected in the production costs and the price required for unchanged economic result (Table 15). Production costs in the modified scenarios were similar for all regions, and both biogas plant sizes. Increased transportation distance due to biogas plant size increase did not have a significant effect. The considerably higher production costs of grass biomass in the L:c scenario compared to the L:m scenario was mainly caused by higher yield-specific costs for machinery and fertilizer. Note, that the costs given for the L:m scenario (112 €/t DM) correspond to an average cost for both feed and biogas substrate production.

Table 15. Grass biomass production costs and price required for unchanged economic result [€/t DM].

Parameter	C1:m	C2:m	L:c	L:m
Production costs				
Feed production			144	112[a]
24 GWh	111	109		
48 GWh	111	110		
Price required for unchanged economic result				
Feed production				112
24 GWh	120	109		
48 GWh	121	111		

[a] Costs for feed and biogas feedstock grass combined.

To have an unchanged economic result in the C1 region, the required price for grass biomass is about 10% higher than the production costs, reflecting a higher profit from the crop that were exchanged for grass. In the C2 region, profits from cereals crops were at the same level as potential profits from grass production, reflected in the fact that costs and required prices were basically identical.

Assuming no necessity to change the internal price of grass fodder, the profit of grass cultivation intensification could be applied to substrate production. Consequently, the price required for grass produced as biogas substrate could be kept low this way, improving the economic sustainability of grass-based biogas production for both farmer and biogas plant.

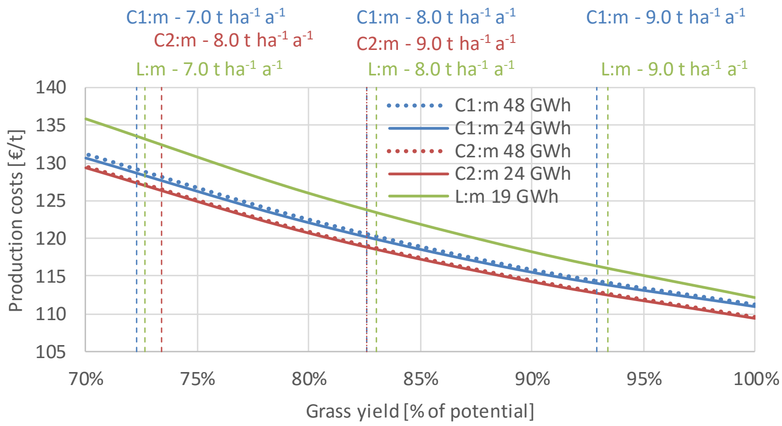

Figure 24. Production costs as depending on grass biomass yields, represented here as yields during full production years.

In cases were actual yields of grass biomass would be lower as assumed in the modified scenarios in the different study regions, production costs per tonne biomass would increase correspondingly (Figure 24). For example, a decrease to 9 t DM/ha would increase substrate costs to approx. 114 and 118 €/t in regions C1 and C2, respectively. In the L region, a decrease in biomass yield to 8 t/ha in the first full production year would result in a substrate cost increase to approx. 123 €/t.

9.5.2 Biogas production

The calculated production cost for biogas from grass crops ranged from approximately 18 – 26 € GJ^{-1} depending on scale, region and feedstock cost as presented in Figure 25. In this analysis, feedstock cost in the C regions represent the calculated production cost as well as the price required in order to maintain the farmer's economic result (Table 15). In the L region, the price is set to 100 € $(t\ DM)^{-1}$. The prices are given per grass DM at harvest, and losses of DM during handling and ensiling (9.5% of DM) were subtracted. In the C regions, feedstock cost had the highest impact on the production cost representing approximately 50 – 55%, excluding cost for distribution.

In C1 as well as in C2, efficiency of scale reduced the production cost with approximately 4% or 1 € GJ^{-1}, although the feedstock cost was slightly higher for the larger biogas plant (172 TJ a^{-1}) due to the increased transport distance.

Adding the estimated cost of 8.9 € GJ^{-1} for distribution of biogas to the calculated production cost results in a required market price of 27 – 35 €/GJ + VAT for private end-users, see Figure 25. Given the estimated market price of 30.7 € GJ^{-1} for private consumers, it is not profitable to produce biogas from grass crops cultivated in the C regions with the assumptions made here.

Reducing the overall production and distribution cost with approximately 2 – 4 € GJ^{-1}, or increasing the market price with the same amount would however be sufficient to reach break-even. Examples of required changes are presented in Table 16.

Given the range of current market price at public filling stations as presented in Appendix F and the range of distribution cost presented in the literature, biogas production from grass crops might however be profitable at certain locations with good conditions. Also, locations with less favorable conditions might require a bigger change of the cost or market structure.

In the L region, the calculated production cost was lower than the estimated market price indicating possible profitability. These calculations are, however, based on the assumption that an existing biogas plant is not fully utilized due to e.g. lack of feedstock. If the additional production of biogas from grass would carry a higher share of the total capital cost, the result would be closer to the one for the other regions. Since the price of the feedstock has such a high impact on the production cost it should also be noted that the price of 100 € ton DM^{-1} should be considered as an example. As presented earlier, the production cost for the grass would e.g. increase with 20 % if the yield dropped from 9 to 8 t ha^{-1}.

It should also be pointed out that the estimated market price is based on a continued tax exemption for biogas which from 2018 requires a 60 % reduction of GHG in new production plants and 50 % in existing. Given the calculated emission reduction presented earlier (Chapter 9.3) this might be possible in the cereal regions but it will probably require special effort.

Table 16. Required changes to reach profitability. 172 GJ a-1 production in region C2 based on grass price required for unchanged result.

Parameter	Calculated (€/GJ)	Required (€/GJ)	Change
Production costs incl. distribution (€/GJ)	32.8	30.7	- 6.8 %
Production cost excl. distribution (€/GJ)	23.9	21.8	- 10 %
Feedstock cost	12.7	10.6	- 20 %
Capital cost	5.7	3.6	- 58 %
Market price biogas	30.7	32.7	+ 6.1%

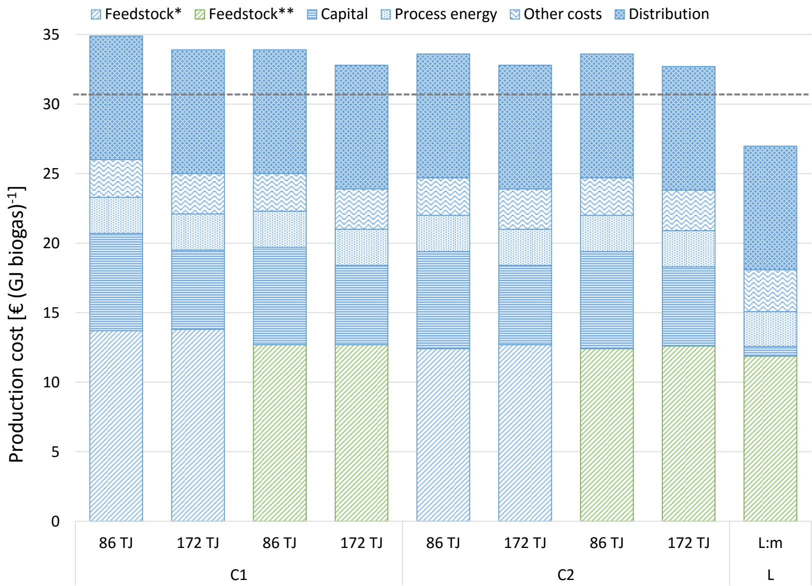

Figure 25: Calculated biogas production cost depending on scale and different feedstock cost
*** Required cost for unchanged economic result for the farmer and ** Production cost. The dotted line represents the current market price for private consumers.**

9.5.3 Socioeconomic value

The reduction in GHG emissions for the modified scenarios compared to the current gives socioeconomic values of 183 (C1) and 187 (C2) € ha^{-1} a^{-1}, when valued according to the current carbon dioxide tax (Figure 26 & Figure 27).

The value of the particle emissions were varied according to where the emissions were assumed to occur (Table 14), and the values in Figure 26 (C1) and Figure 27 (C2) represent assumptions where all emissions occurred in the countryside (lowest values) or all emissions in production occurred in the countryside, but the biofuel utilization reduced in smaller Swedish towns (highest value).

The eutrophication and acidification potentials were both increased for the investigated modified scenarios, which gave a negative socioeconomic impact. Applying the range of values shown in Table 14 will give the range of negative socioeconomic impacts shown in Figure 26 (C1) and Figure 27 (C2).

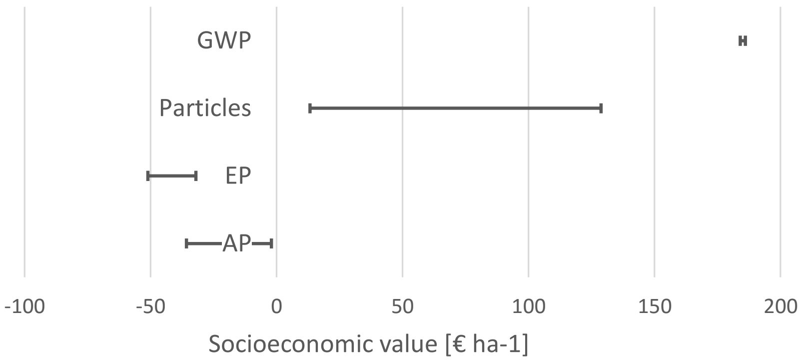

Figure 26. The variation in socioeconomic value for the investigated environmental impacts in region C1.

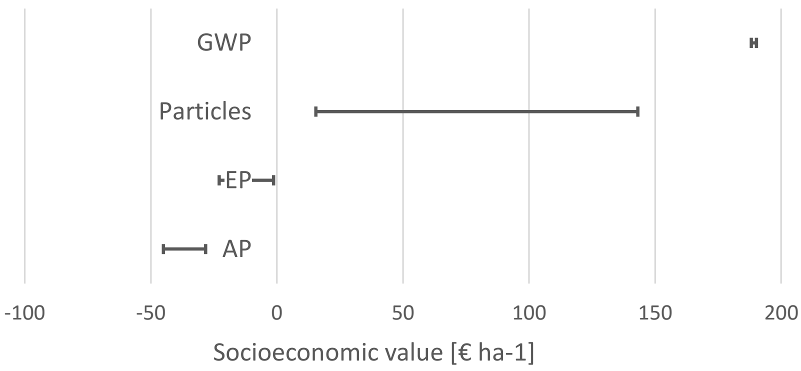

Figure 27. The variation in socioeconomic value for the investigated environmental impacts in region C2.

The total socioeconomic value, given the positive and negative contributions shown in Figure 26 and Figure 27, ranges from 160-230 € ha^{-1} a^{-1} (C1) and 180-260 € ha^{-1} a^{-1} (C2) depending on if lowest or highest values for EP, AP and particle emissions are used. The value for avoided greenhouse gas emissions is not varied, and applying the higher value suggested for sensitivity analyses by Trafikverket (2016) (not shown in figures) would increase the socioeconomic value significantly, to 5-600 € ha^{-1} a^{-1}.

To enable comparison with the values calculated in the previous sections, the socioeconomic value was recalculated per amount of harvested grass DM. In Figure 28, an example for region C2 is shown, where the required price for unchanged economic result for the farmer is compared to the price that the biogas plant (172 TJ a^{-1}) can pay for grass as biogas feedstock at present gas prices (Table 16). The socioeconomic value of 260 € ha^{-1} a^{-1} (assigning EP, AP and particle emissions the highest values in the range evaluated) is recalculated per t of harvested grass DM (3.0 t DM ha^{-1} a^{-1} as average for the crop rotation).

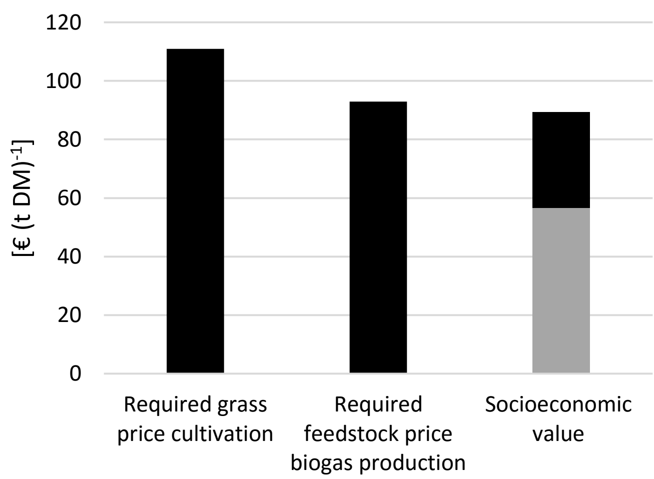

Figure 28. Comparison of socioeconomic value with other economic results on the basis of harvested grass DM. The grey bar shows the value of the present exemption from CO_2-tax for biogas used as vehicle fuel (for natural gas used in the same application, a CO_2-tax of 0.26 € m^3 is applied (Skatteverket, 2016).

The comparison in Figure 28 shows that it could be motivated from a socioeconomic point of view to cover the gap between the required price for the farmer and the price that can be paid by the biogas plant, even when the negative impact on emissions of acidifying and eutrophying compounds is taken into account. The utilization of biogas as vehicle fuel is, however, already exempted from CO_2-tax until 2020 (Figure 28, grey bar). This exemption would today be granted biogas from grass that fulfils the required reduction of greenhouse gas emission (calculated according to EU RED) of 60% (new installations) or 35% (existing plants). It is today unclear what kind of policy measure that will be applied for biogas from grass cultivated on arable land, and for biofuels in general, after 2020.

10 Conclusions

In both the evaluated regions with cereal-based crop rotations, soil organic carbon is lost each year. From a soil quality perspective, this development is not sustainable in the long run, and also turns arable land into a source of CO_2 emissions. Stopping and even reversing this trend by transforming arable land into a carbon sink requires powerful measures. The approach investigated here was to maximize carbon input in highly productive crop rotations by cultivating grass two out of six years, using the grass for biogas production and recirculating the produced biofertilizer in the crop rotation. This was shown to be a measure that could stop and even reverse the carbon loss, which apart from soil quality aspects gave a large benefit when analysed from a climate perspective. If applied on the 274 000 ha of arable land identified in the two regions, emission of 0.2 Mt CO_2-eq per year would be avoided. The produced biogas would give another 0.2 Mt CO_2-eq per year emission reduction when replacing fossil fuels for transport. This could be compared to the total annual emissions of 10 Mt CO_2-eq from the agricultural sector (SEPA, 2015).

At the same time, the increased use of biofertilizer in the modified scenarios increased the ammonia emission to air, impacting both acidification and eutrophication. Awareness of this conflicting environmental impact can only be created by applying a broader systems perspective. With this negative impact taken into consideration, the total socioeconomic value of the investigated modification was still shown to be positive.

The economic sustainability in agriculture was investigated through calculation of a required price for grass to maintain the same revenue as for the current cereal based crop rotations. This price was shown to be too high to allow profitable biogas production in dedicated biogas plants given average market conditions as estimated here. The economic viability of grass production in these regions needs to be improved if grass is to be used as a tool for carbon sequestration in respect to improving food production, and the calculated socioeconomic value shows that it can be beneficial to bridge the gap between grass production costs and acceptable biogas feedstock costs. A minor increase in market price or similar cost reduction in production and distribution would be enough to reach profitability. In some cost efficient biogas systems, it might even be so today.

A prerequisite for economic viability is the subsidy through exemption from the CO_2 tax, which is applied until 2020 for biogas as automotive fuel under the condition that the biofuel fulfils the present sustainability criteria in EU RED. For new installations, this means a reduction in greenhouse gas emissions of 60% compared to fossil fuels, calculated according to the methodology of the EU RED. According to this methodology, the carbon sequestration impact of the grass is to be excluded, and consequently, the climate benefit is only barely enough to fulfil the requirements.

For comparison, the same calculations were made for an alternative scenario outlined for a livestock-dominated region. Here, grass produced as biogas feedstock would fulfil the sustainability criteria in EU RED, and is economic viable, which makes it an interesting option in this region. However, the livestock region would be the region to benefit least in increased soil quality.

This illustrates how the regional differences in carbon sequestration potential are overlooked, both in economic evaluations and when climate impacts are valued according to the present EU sustainability criteria. Future sustainability assessments for arable land use should include the dimension of soil carbon increase and decrease, e.g. by including a soil-specific bonus for crops contributing to increasing soil organic carbon.

The complexity in sustainability assessments of different uses of arable land requires an improved methodological framework. In the discussion of future shortages in food/feed production arise restrictions within the EU for the use of arable land for energy crop production. On the other hand, current food production systems may not be sustainable as demonstrated with the regional loss of soil organic carbon. It is important to broaden the perspective, and to make sufficiently wide-ranging analyses of such complex systems as the use of arable land, for example by taking the aspect of converting crop rotations as a whole to carbon sinks into consideration. Scientifically sound assessments at a national level are thus important in future policies regarding sustainable use of arable land. The work on the EU bioenergy policy after 2020 is presently in progress. The results of the present study stresses the importance of taking local conditions and spatial perspectives into account to avoid counterproductive measures when sustainability criteria are to be formulated.

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Appendix A. Regional differences in agriculture

A.1 SELECTION OF STUDY REGIONS BASED ON REGIONAL CHARACTERISTICS

Within the regions of Skåne and Västra Götaland there are cultivation areas characterized by dominating cereal production and low livestock density. This production pattern usually means uniform crop rotations and low availability of biofertilizers in the form of manure as well as marginal grass-clover production as feed crop. This can lead to problems with low or decreasing soil organic carbon contents and soil compaction. It was specifically for this type of region we wanted to perform our assessments. Here, we wanted to examine the potential environmental gains, positive effects on soil organic carbon content and contribution with biomass as a biogas substrate from integration of grass production into the crop rotation.

For comparison, we chose a region characterized by opposite conditions, i.e. a region with a limited area for cereal cultivation, a high share of grass (-clover) feed crops and a high livestock density.

A.1.1 Methodology

For choosing interesting study regions typical for cereal-dominated crop rotations and those with a high share of grass-clover crops, data aggregated on the basis of *harvest area* (skördeområde, SKO) was used (Figure 29). On this level, much data is available and the data resolution is high enough in order to prepare a sound basis for selection. However, data on livestock production (i.e. livestock units, pasture area) was available only as aggregated on municipality level (Figure 29).

Note, that for the further assessment of the chosen regions, data resolution on the basis of *harvest areas* (skördeområden, SKO) was used for normal biomass harvest levels, since these are only reported on this level. Figure 29 shows the borders of two different data resolutions.

For choosing regions representative for the situation described above, we have calculated the share of land used for cereals and the share of land used for grass crops in SKO. Data on the land area used for cultivation of cereal and grass crops and data on the number of livestock units kept in each municipality were provided from the Board of Agriculture (Olsson, 2015; SJV, 2015d).

The share of land used for cereal production was calculated as sum of the area used for cultivation of winter and spring wheat, rye, winter and spring barley, oat, mixed grain and triticale. The share of land used for grass crop production was extracted as the area used for cultivation of temporary grass (*slåttervall*). In order to reflect the current situation, but minimizing the risk for outliers, the data was aggregated as averages for the year 2009-2013, i.e. a 5-year period.

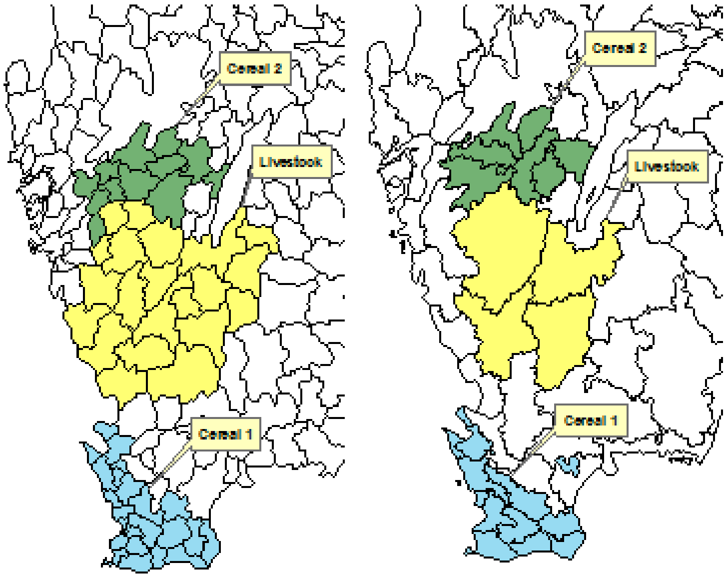


Figure 29. Comparison of borders for municipalities (left) and harvest areas (skördeområden, SKO, right).

A.1.2 Historic development

Agricultural land use has gone through many structural changes the last century. In Sweden, arable land area peaked in the 1920ies with 3.8 million ha, and has since then decreased steadily, in 2014 to below 2.6 million ha (Figure 30) (SJV, 2011, 2016b)

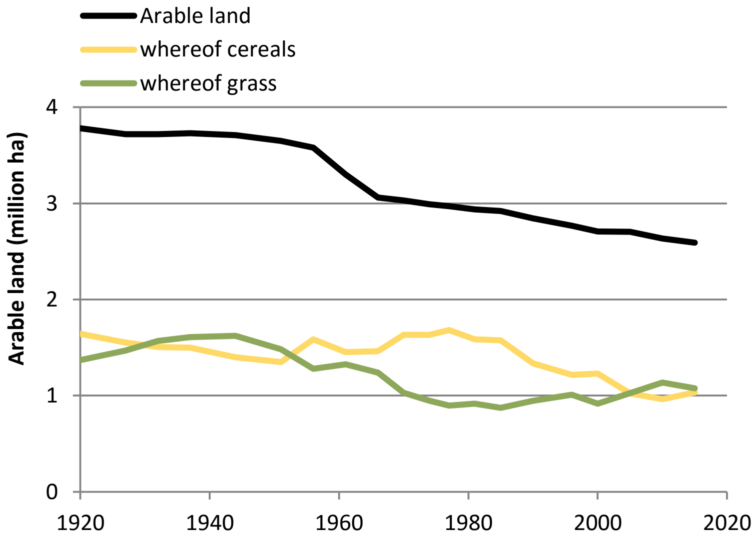


Figure 30. Historic development of the area of arable land in Sweden (SJV, 2011, 2016b)

The two dominating crop types in 1920 and today were grass-clover crops and cereals. While the total area of arable land use decreased, area used for cereal production remained rather constant at around 1.5 million ha until the 1980ies where after it decreased to presently around 1 million ha. In the 1930-40ies, grass-clover crops were cultivated on almost half of the arable land, but since then steadily declined to today around 1 million ha (SJV, 2011, 2016b). The regional differences are, however, large.

The livestock numbers within Swedish agriculture have been steadily decreasing. As an example, the number of cattle peaked in 1930-ies with 3 million, and since then decreased to about half (SJV, 2011).

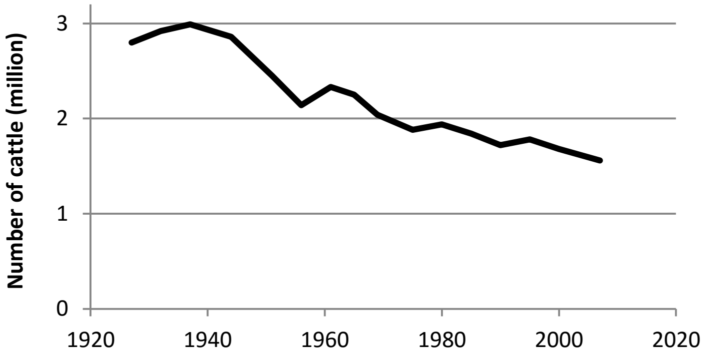


Figure 31. Historic development of cattle numbers in Sweden (SJV, 2011).

A.2 PASTURE LAND AND ARABLE LAND

The chosen cereal regions C1 and C2 are further characterized by relatively large ratios between arable and pasture land (Figure 32), indicating a low potential to produce grass crops from pasture land.

In contrast, the livestock region L, is characterized by an arable/pasture area ration of below 5. Note, that the livestock assessment is based on data aggregated on municipality level. Higher arable/pasture area ratios can be found in coastal municipalities. These areas are however included in the livestock region as defined by harvest areas (skördeområden, SKO).

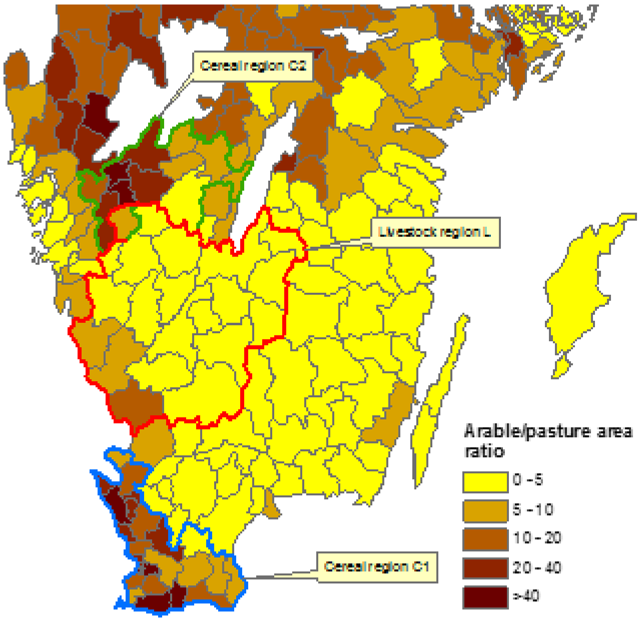


Figure 32. Ratio between arable land area and land area under pasture based on data average for years 2009-2013, based on data from (SJV, 2015d).

A.3 CEREAL REGIONS C1 & C2

A.3.1 Current cereal crop rotations

For each region typical crop rotations were chosen. These crop rotations were adjusted from the results of an extensive study analysing the cultivation area of crops and the most probable corresponding pre- and post-crops covering a number of agricultural production areas in Sweden (SJV, 2006).

Table 17. Crop rotations typical for the chosen study regions C1 and C2 (SJV, 2006) and modified crop rotations including grass crops.

	Cereal region C1	Cereal region C2
	C1:c (current)[a]	**C2:c (current)[a]**
Year 1	Winter wheat	Winter wheat
Year 2	Sugar beets	Oat
Year 3	Spring barley[b]	Winter wheat
Year 4	Winter wheat	Winter wheat
Year 5	Spring barley[b]	Spring barley[b]
Year 6	Winter oilseed rape	Oat

[a] Based on suggestions by SJV (2006).
[b] Malting barley

In order to investigate if significant changes in the crop rotations may have occurred since the period investigated in this study (1996-2004), development of area of set-aside land, cultivation area of winter wheat and cultivation area of oil crops (oilseed rape and turnip rape) has been investigated.

For this purpose, data aggregated on municipality level was used, where data was available for the data period of the study (1996-2004) as well as data from 2014 (SJV, 2015b).

For study region C2, one year of set-aside was originally included in the 7-year crop rotation (year 3). However, the area set-aside from cultivation has decreased significantly, both in region C1 (86% decrease on average) and in region C2 (52% decrease on average), Figure 33. Therefore, the suggested set-aside year in the crop rotation for study region C2 was removed from the crop rotation.

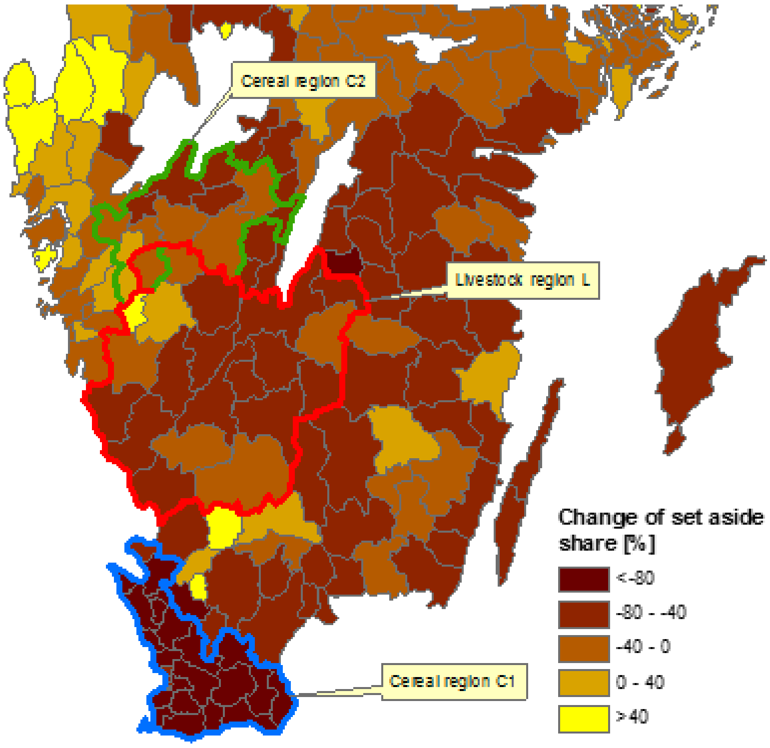


Figure 33. Change of set-aside land area calculated as relative change (%) between the mean during the data inventory period 1996-2004 and the current situation (2009-2013).

The share of cereal cultivation area has decreased only slightly between the inventory period (1996-2004) and the current situation (2009-2013) in both cereal regions C1 (5 % decrease on average) and C2 (20% decrease on average) (Figure 34). This confirmed the dominance of cereals in the cereal regions C1 and C2 and supported the crop rotations as presented in Table 17.

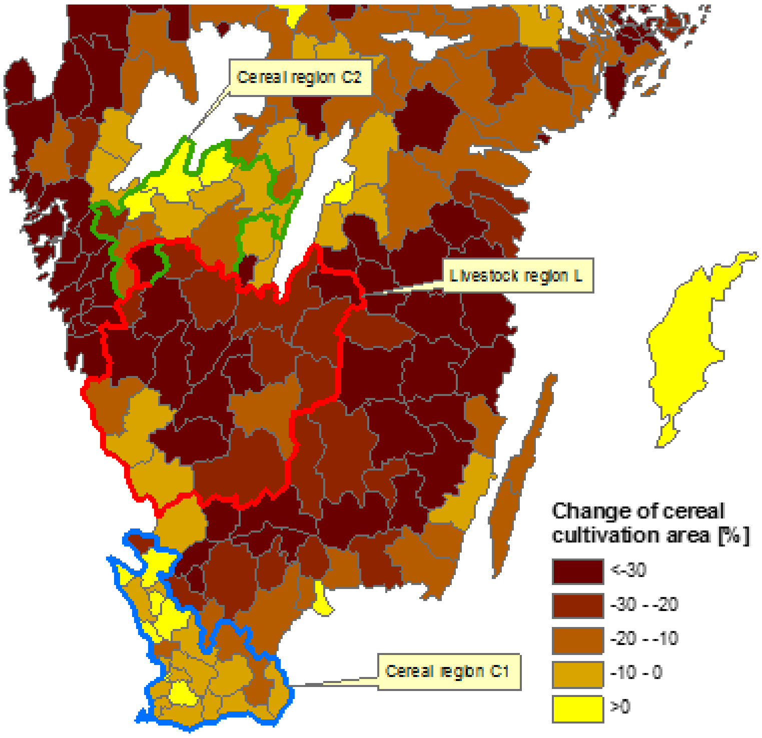


Figure 34. Change of share of winter wheat cultivation area calculated as relative change (%) between the mean during the data inventory period 1996-2004 and the situation during 2014.

The share of *Brassica* oil crop (oilseed and turnip rape) cultivation area has increased significantly in region C1 (Figure 35). In some municipalities in study region C1, the share of *Brassica* oil crops approached the frequency of 15%, i.e. cultivation every 7th year. However, the increase is not assumed to change the crop rotation as presented in Table 17.

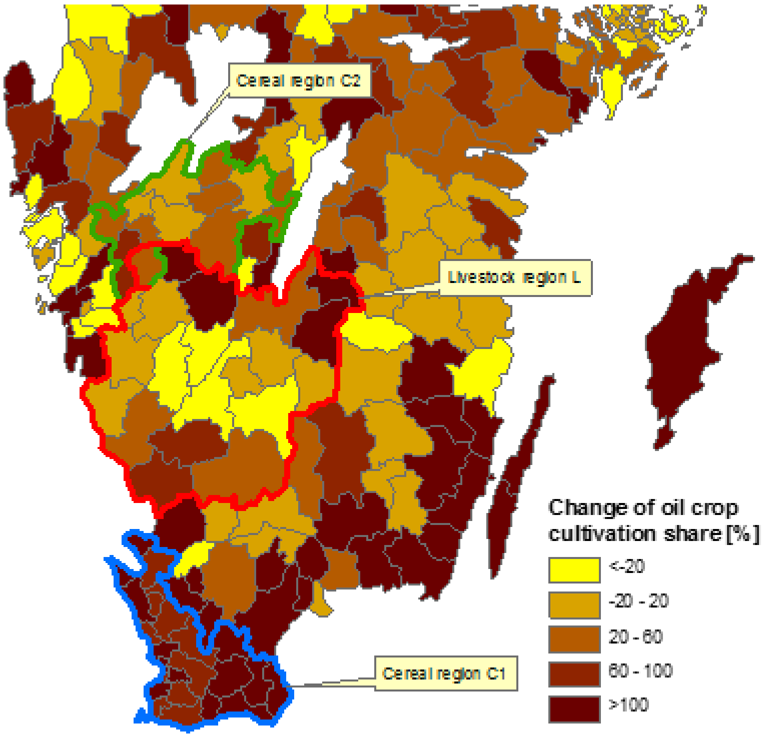


Figure 35. Change of share of Brassica oil crop (oilseed and turnip rape) cultivation area calculated as relative change (%) between the mean during the data inventory period 1996-2004 and the situation during 2014.

A.3.2 Modified cereal crop rotations

For the two cereal regions, alternative crop rotations including two years of grass crops were chosen (Table 18). This was done by exchanging one year of spring barley and one year of oilseed rape for grass crops in cereal region C1. In cereal region C2, one year of spring barley and one year of oat were exchanged for grass crops.

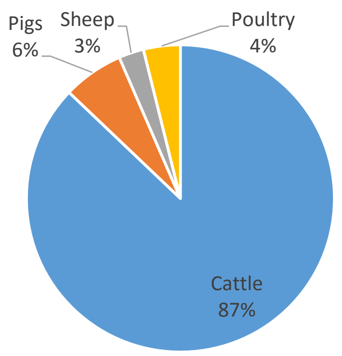
Table 18. Modified crop rotations for cereal regions C1 and C2.

	Cereal region C1	Cereal region C2
	C1:m (modified)	C2:m (modified)
Year 1	Winter wheat	Winter wheat
Year 2	Sugar beets	Oat
Year 3	Spring barley[b]	Winter wheat
Year 4	Winter wheat	Winter wheat
Year 5	**Grass, year I**	**Grass, year I**
Year 6	**Grass, year II**	**Grass, year II**

[b] Malting barley

A.4 LIVESTOCK REGION L

The livestock production region L is dominated by cattle husbandry for both milk and meat production (Figure 36, left). It is further characterized by a substantially higher cattle density compared to the cereal study regions (Figure 36, right, (SJV, 2015b)).



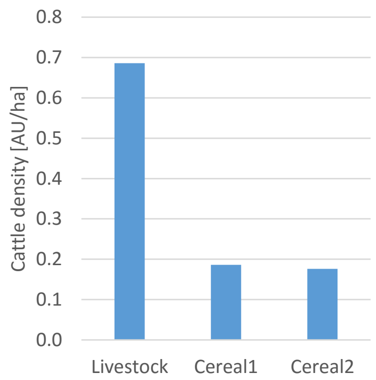


Figure 36. Livestock production in the livestock region (left) and cattle density in all study areas (right).

In the Livestock region, grass-clover blends are cultivated as coarse animal feed for production of milk, meat, but also as feed for horses and sheep. The two main agricultural focus areas are meat production and milk production[8].

A.4.1 Meat production

Meat production farms produce grass-clover crops used as coarse feed, but usually with a lower production intensity compared to milk farms. The grass-clover crops are typically harvested 2 times per year at yields around 6 (3+3) t DM/ha/year or the fields are harvested only once and then used for grazing[9]. Outside crop rotations, there is grass-clover production where the crops are cultivated for more than 5 years, typically 7-8 years, before the soil is ploughed again and new grass-clover crops are established. This is usually carried out on small lots of arable land (1-1.5 ha). These longer grass-clover cultivations outside crop rotations are usually used by smaller meat-producing farms. Meat-producing farms have on average 36 hectares of arable land at their disposition (SCB, 2014a). Meat farmers often calculate the amount of feed necessary and when this amount is produced cultivation intensity is reduced for the remaining season. This demand-regulated feed production may open up for intensification of grass-clover production.

A.4.2 Milk production

The dominating application of grass-based feed is milk production. Typical crop rotations on milk farms usually consist of only one year of whole-crop cereals used as concentrates and 3-4 years of grass-clover crops. Farms that have a generous land area available may have a second year of whole-crop cereals and 3-4 years of grass-clover crops. Milk-producing farms have on average 105 hectares of arable land at their disposition (SCB, 2014a).

[8] Ola Hallin, HS Sjuhärad. Personal communication October 2015
[9] Ola Hallin, HS Sjuhärad. Personal communication October 2015

All harvested biomass remains and is utilized on the farm, i.e. no sale of feed materials occurs. Cultivation lots are rather small, 1-1.5 ha. Grass-clover crops are cut three times a year and are fertilized with liquid manure and additional nitrogen fertilizer of up to 140 kg N/ha. This yields 7-8 t DM/ha/year of high quality feed, which removes about 200 kg/ha of potassium. Since fields in the region a typically moraine soil, there is little potassium added from bedrock weathering processes, which renders the manure an important potassium fertilizer. The grass-clover crops are typically stored as round bales. Only larger farms own bunker silos.

A.4.3 Current livestock crop rotation

In the current Livestock scenario, it is assumed that oat and grass-clover crops are cultivated on a milk farm (Table 19), with grass-clover crops being cultivated for three years in a row. The grass crops are cut three times per year and fertilized with manure complemented with mineral fertilizer, resulting an average annual biomass yield of 6.5 t DM/ha, which can be compared with the harvest level of 5.4 t DM/ha typical for the region (SCB, 2014d). Oat is grown in one out the four years of crop rotation.

Table 19. Current typical crop rotation in the Livestock region.

	Livestock region L
	L:c (current)[a]
Year 1	Oat
Year 2	Grass, year I
Year 3	Grass, year II
Year 4	Grass, year III

[a] Based on suggestions by SJV (2006).

A.4.4 Modified livestock crop rotation

In the modified Livestock scenario, the same crop rotation is assumed as in the current livestock scenario. In contrast, grass crop cultivation is assumed to be intensified, resulting in higher DM yields (0).

Appendix B. Crop production

B.1 CROP YIELDS

B.1.1 Grass crops

The principles for grass sowing and harvest are described in Chapter 6.3.1.

For grass crops, normal harvest level data for high-intensity production was not available, since official statistics include even low and medium-intensity production systems, as well as organic production systems (SCB, 2014c). Therefore, expectable biomass yields have been estimated based on variety field experiments hosted by the Field Research Unit (FFE) at the Swedish University of Agricultural Sciences.

Biomass DM yields were based on a grass crop mix of perennial rye-grass, meadow fescue and timothy-grass, which were assumed to represent 40, 30 and 30% of the biomass in the field. This mix of grass species was deemed to result in stable yields when harvested three times per year, over the course of two to three consecutive years according to recommendations from the Swedish Board of Agriculture. No biomass yield was attributed to legumes such as red and white clover, since the high nitrogen fertilization levels expected in intensively cultivation grass-clover crops will lead to grasses outcompeting legumes. Earlier studies have shown that the share of legume biomass in intensively cultivation grass-clover crops is negligible (Prade *et al.*, 2015; Mårtensson *et al.*, submitted). Variety experiments for the above mentioned grass species were chosen, since in these experiments grass species are cultivated in a way to demonstrate the biomass yield potential and are fertilized with high rates of nitrogen, potassium and phosphorous, so that availability of plant nutrients will not limit plant growth.

For each grass species, variety experiment results available from FFE were extracted for the period of 2004-2014 for each harvest occasion for field experiments harvested 3 times per year and two, two and three consecutive years for regions C1, C2 and L, respectively (FFE, 2016).

For all regions, mean share of each harvest occasion was calculated by dividing the sum of biomass yields from each harvest occasion for all found field experiments by the sum of the total annual biomass yield.

Yield differences between regions

The analysis of differences in biomass yield between regions was carried out for each variety of the whole set of found experiments in order to avoid inter-variety differences. Harvest differences due to annual changes in pedoclimatic conditions could not be excluded, but the effect of such differences were assumed to be negligible due to the large number of field experiments used for each dataset. Differences were calculated from the slope of a regression line (forced through 0) for all pairs of biomass yields for the C1 and C2 region as well as for the C1 and L region.

Absolute biomass yields

For the region C1, absolute biomass yields were then calculated for each grass-clover crop species using the mean biomass yield across all varieties for the region C1. For regions C2 and L, absolute biomass yields were calculated from the C1 biomass yields and the relative difference between C1 and the other regions. Biomass yield results from variety experiments are, however, harvested at cutting heights low to the ground (4-6 cm stubbles) and therefore need to be adjusted to represent harvest by standard field machinery (e.g. forage harvester). For adjustment, recovery coefficients were calculated for each of the three harvest occasions based on the following empirical relation (Prade *et al.*, 2015).

$$\text{Recovery coefficient }[\%]=1.3828\cdot\text{Biomass yield }\left[\frac{\text{t}}{\text{ha}}\right]+64.603$$

The corrected biomass yields for each harvest occasion were then summed up to receive the total biomass yield (Table 20).

Table 20. Calculated biomass yields for grass-clover crops in the study regions.

Region	Biomass DM yield [kg ha^{-1}]										
	Year 1				Year 2				Year 3		
Cut	I	II	III	Total	I	II	III	Total	I	II	Total
C1 modified	4559	2826	2301	**9686**	4317	2000		**6317**			
C2 modified	5288	2780	2829	**10897**	4526	2264		**6791**			
A current	3352	1861	1944	**7157**	3151	1750	1691	**6592**	1953	2164	**4117**
A modified	4513	2505	2617	**9634**	4242	2356	2276	**8874**	2629	2913	**5543**

B.1.2 Other crops

Crop yields for the food and feed crops of the crop rotations presented above, with the exception of grass-clover crops, were taken from official statistical sources in the form of standard biomass yields (SCB, 2014d). Precrop effects for winter oilseed rape, grass, oat and sugar beets were taken into account for the final yields in the crop rotations following recommendations by the Swedish Board of Agriculture (Albertsson *et al.*, 2015).

Table 21. Standard biomass dry matter yields [kg ha^{-1}] for crops grown in the studied regions (SCB, 2014d).

Crop	Study region		
	C1	C2	L
Winter wheat	6254	4733	
Sugar beets	12884		
Spring barley	4562[a]	3940[a]	
Winter oilseed rape	3527		
Oat	4203	3619	2994

[a] Malting barley

B.2 CROP FERTILIZATION

Amounts of plant nutrients required for crop fertilization at corresponding biomass yields was carried out based on official recommendations for nitrogen and based on typical biomass nutrient contents for phosphorous and potassium (Table 22). Recommendations for nitrogen applications (SJV, 2014) were translated into a basic and a yield-corresponding amount using a simple linear regression model, except for sugar beets where a fixed nitrogen level of 120 kg N/ha was assumed. In this model, biomass nutrient content was assumed to be universal for all regions, but the basic nitrogen amount varied between regions.

Table 22. Parameters for calculation of plant nutrient amounts for crop fertilization.

Crops	Base C1 & L	Base C2	Biomass nutrient content		
	N[a]	N[a]	N[a]	P[b]	K[b]
	[kg ha^{-1}]	[[kg ha^{-1}]	[% of DM]	[% of DM]	[% of DM]
Grass	53	53	2,00	0.22	2.10
Oat	-9.5	0.5	1.92	0.38	0.50
Spring barley	15	20	1.74	0.40	0.50
Sugar beets	120			0.18	0.91
Winter oilseed rape	65		2.20	0.66	0.88
Winter wheat	32	42	1.86	0.36	0.50

[a] Calculated from official recommendations (SJV, 2014)
[b] Calculated from SJV (SJV, 2010) using moisture contents of 0 % (grass), 14 % (cereals), 9 % (oilcrops) and 78 % (sugarbeets).

For the above regression model, the amounts of plant nutrients as required by the corresponding biomass yields were calculated (Table 23).

Table 23. Plant nutrient demand [kg ha^{-1} a^{-1}] according to crop yields assumed in the different scenarios.

	C1:c	C1:m	C2:c	C2:m	L:c	L:m
Year 1	Winter wheat	Winter wheat	Winter wheat	Winter wheat	Oat	Oat
N	120	130	134	114	72	72
P	25	22	18	17	12	12
K	34	30	25	23	15	15
Year 2	Sugar beets	Sugar beets	Oat	Oat	Grass, year I	Grass, year I
N	120	120	83	83	196	246
P	24	24	14	14	16	21
K	118	118	18	18	150	202
Year 3	Spring barley	Spring barley	Winter wheat	Winter wheat	Grass, year II	Grass, year II
N	83	83	134	136	185	230
P	20	20	18	18	15	20
K	25	25	25	25	138	186
Year 4	Winter wheat	Winter wheat	Winter wheat	Winter wheat	Grass, year III	Grass, year III
N	137	137	121	121	135	164
P	20	20	15	15	9	12
K	28	28	21	21	86	116
Year 5	Spring barley	Grass, year I	Spring barley	Grass, year I		
N	90	247	89	271		
P	17	21	16	24		
K	21	203	20	229		
Year 6	W oilseed rape	Grass, year II	Oat	Grass, year II		
N	143	179	83	189		
P	23	14	14	15		
K	31	133	18	143		

The amounts of nutrients added in form of biofertilizer in Scenarios C1:m, C2:m, L:c and L:m, and the corresponding decrease in mineral fertilizer demand for the scenarios are shown in Table 24.

Table 24. Amounts of plant nutrients [kg ha^{-1} a^{-1}] applied in crop production in the form of mineral fertilizer and biofertilizer. N in biofertilizer given as total ammoniacal nitrogen (TAN) before field losses, which are assumed to be 3% of added TAN in winter wheat and 20% of TAN in grass.

	C1:m[1]		C2:m[1]		L:c[2]		L:m[2]	
	Mineral fertilizer	Bio-fertilizer	Mineral fertilizer	Bio-fertilizer	Mineral fertilizer	Bio-fertilizer	Mineral fertilizer	Bio-fertilizer
Year 1	Winter wheat		Winter wheat		Oat		Oat	
N	69	70	40	75	72	-	72	-
P	11	12	4	13	12	-	12	-
K	-32	64	-45	69	15	-	15	-
Year 2	Sugar beets		Oat		Grass, year I		Grass, year I	
N	120	-	83	-	162	44	155	113
P	24	-	14	-	-2	17	-4	25
K	118	-	18	-	52	98	63	139
Year 3	Spring barley		Winter wheat		Grass, year II		Grass, year II	
N	83	0	71	65	150	44	140	113
P	20	0	6	11	-3	17	-6	25
K	25	0	-35	60	40	98	47	139
Year 4	Winter wheat		Winter wheat		Grass, year III		Grass, year III	
N	78	68	61	62	101	44	73	113
P	10	12	4	11	-8	17	-13	25
K	-32	64	-35	57	-12	98	-22	139
Year 5	Grass, year I		Grass, year I					
N	157	112	172	123				
P	2	20	2	22				
K	101	102	116	112				
Year 6	Grass, year II		Grass, year II					
N	125	68	168	26				
P	2	12	10	5				
K	71	62	119	24				

[1] Spring barley, oat and sugar beets received all nitrogen in one application, while winter oilseed rape and winter wheat received nitrogen in three applications. Grass crops received a first application in spring and an additional application after the first and second harvest or only after the first harvest in a full production and break year, respectively.
[2] Oats received mineral fertilizer in one application in spring. Grass crops received a start application in spring and an additional application after the first and second harvest or only after the first harvest in a full production and break year, respectively.

B.3 GRASS PROPERTIES

The composition of the biomass used for biogas production influences aspects like biodegradability and contribution of nutrients and carbon via the recycled digestate. Grass is, however, not one plant type, so apart from the variation that will occur based on cultivation aspects, different mixes of species/varieties will influence composition. For the present study, data from own and other studies was summarized as a basis for selecting input data on representative grass properties, and to investigate if and how composition could be linked to the length of the growth period. Also, the analyses commonly performed for grass (for forage) are not always the same as the parameters that are relevant to define when the grass is to be used as a biofuel feedstock. Therefore, a short summary of typical and available analyses is included, and how they were used in the present study. Finally, the composition used in the study and the compositional changes that will occur due to biochemical degradation during storage and handling of the grass are described.

B.3.1 Commonly assessed parameters

The content of dry matter (DM) is a common parameter to quantify, both when grass is used as forage, and for biogas production. The composition is thus presented on a DM basis. DM is normally analysed by oven drying (APHA, 2005). Important to consider is that if the sample contains volatile substances like organic acids and alcohols, which is the case for ensiled grass, DM determination should be performed with a correction for the loss of volatiles, as suggested by e.g. Porter and Murray (Porter and Murray, 2001). Failing to do so will cause error in DM determination, which could lead to wrongful interpretations of e.g. DM losses during ensiling (Kreuger *et al.*, 2011). In the context of biogas production, volatile solids (VS) is a commonly used concept. Volatility in this case does not refer to compounds that are evaporating, but to compounds that are oxidized at high temperature (550-650^{o}C) (Sluiter *et al.*, 2008). VS is used as a rough measure of the content of organic material, and the remaining solids is called ash.

The VS-fraction can be further divided in groups of compounds, and common in forage and food analyses is the determination of crude protein, crude fat and a residual fraction which is called "fibre (Cherney, 2000; Pond *et al.*, 2005). The residual fraction can be further subdivided into the groups Acid Detergent Fiber (ADF) and Neutral Detergent Fiber (NDF), where ADF gives an approximation of cellulose and lignin content, and NDF also includes hemicellulose (Van Soest and Wine, 1967). These analyses do not include detailed determinations of individual compounds, but are defined as proximate analyses (Björnsson *et al.*, 2014).

More detailed analyses can be performed to determine the content of lignin and carbohydrates, including structural carbohydrates like cellulose and hemicellulose, and water soluble carbohydrates (WSC) (Sluiter *et al.*, 2005; Sluiter and Sluiter, 2005; Sluiter *et al.*, 2011). These analyses are common for lignocellulosic biomass to be used for the production of ethanol and other biofuels, and allows theoretical calculations of yields.

In the determination of methane production, a detailed information on the composition of carbohydrate and other compounds can be used for calculation of the theoretical yields. Such a calculation can also be performed based on information from proximate analyses, where a typical chemical composition for a group like proteins and lipids then has to be assumed (Björnsson *et al.*, 2014). A theoretical methane yield is good to have as a reference value in addition to the commonly used experimental determination of methane yields, the biochemical methane potential (BMP) determination e.g. (Angelidaki *et al.*, 2009). The BMP will give the methane yield as determined in a batch experiment under laboratory conditions, often with a biomass sample that is very finely cut, and under optimal conditions. Assumptions then have to be made on how high the practical full scale methane yield can be in relation to BMP.

B.3.2 Grass properties

Swedish studies published in the period 1986-2013 with analyses of grass and grass-clover for forage from south to mid-Sweden with different number of harvests and harvest dates has been summarized by Gunnarsson *et al.* (2014). Data on content of NDF and crude protein is presented, and is shown in Figure 37 in relation to the length of the growth period. In addition, ash content in relation to length of growth period for other grass and grass-clover samples from cultivation trials in southern Sweden are shown (Gissén *et al.*, 2014; Prade *et al.*, 2015). There is no obvious change in relation to length of growth period for any of these components within this interval, the average for all values shown in Figure 37 are 50%, 17% and 9% of DM for NDF, crude protein and ash respectively.

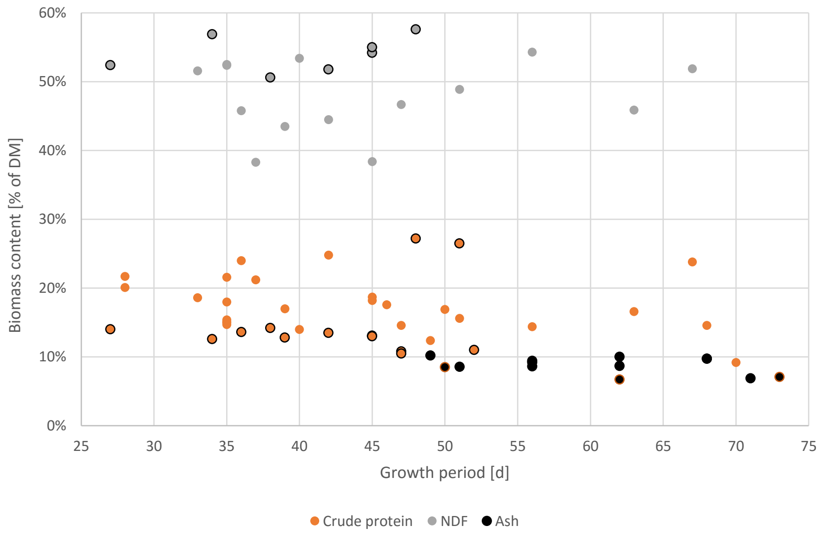


Figure 37. Biomass content of NDF, crude protein and ash depending on number of growth days for the crop. The values for NDF and crude protein with a black circle are for the first cut in May-June where a start date for growth was not given, and is based on an assumed start date of 30th of April. Values for ash content with an orange circle are own data that is previously unpublished.

As comparison, data from a British study is shown, where similar components in relation to number of growth days is evaluated for Ryegrass (McDonald *et al.*, 1991) (Figure 38). In this evaluation of long periods of growth, clear compositional changes can be seen, with increasing fibre content (lignin, cellulose, hemicellulose) and decreasing content of easily biodegradable components (WSC, crude protein) with increasing growth period.

In the present study, the grass is harvested frequently and the growth period ranges from 42-56 days. Based on the data shown in Figure 37, the composition is assumed to remain constant within this interval.

More detailed data on grass composition than what is shown in Figure 37 was desired to enable theoretic calculations of biodegradability and changes during the handling and storage of the grass. Detailed data on composition, including structural carbohydrates, is more common in relation to biomass commonly investigated as bioethanol feedstock, and the majority of the data found was for individual grasses like Timothy (*Phleum pretense*), Reed canary grass (*Phalaris arundinacea L.*) and Switchgrass (*Panicum virgatum*). Data on the content of protein and fat, which is common to find for grass used as forage, was then not available (Björnsson *et al.*, 2014). Only two studies where the biomass was described as grass or a mix of grass for silage production, and where all the desired compounds were reported, were found (Table 25).

Table 25. Compositional analysis of grass

Source	*Ash*	*Lignin*	*Cellulose*	*Hemi-cellulose*	*Non-structural carbohydrates*	*Lipids*	*Crude protein*
	% of DM						
(Koch *et al.*, 2010)	8.8	5.3	27	22	17	3.0	17
(Carlsson *et al.*, 2013)	9.9	18	15	13	26	1.7	17
Mean	**9.4**	**12**	**21**	**17**	**21**	**2.4**	**17**

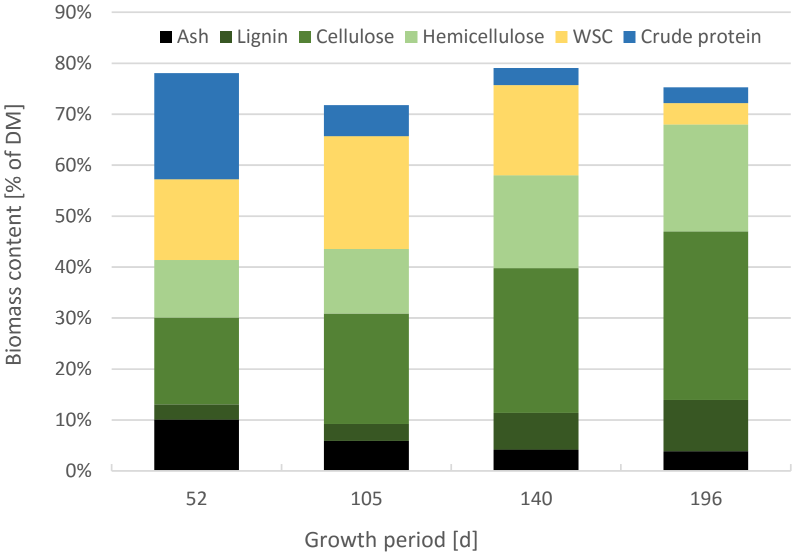


Figure 38. Compositional analysis from a British study on Ryegrass. Based on data from McDonald et al (McDonald *et al.*, 1991) with an assumed start of growth 1 March. WSC=water soluble carbohydrates.

The mean values for these two analyses compare well to the data shown in Figure 37. The average protein content of 17% is the same as the mean value, the lignin, cellulose and hemicellulose sum up to 50% of DM, which is also the NDF mean value in Figure 37, and the ash content is 9% in both cases. The data on non-structural carbohydrates (21% of DM) is similar to values presented by McDonald et al. (McDonald *et al.*, 1991) for Ryegrass (16-22% of DM) and Timothy (20% of DM) at shorter growth periods (harvest from April to early June). Values presented in a Swedish data compilation are lower (Ryegrass 16% of DM, Timothy 10% of DM), but in this latter study no information on growth period is given, and given values could be mean values from a wider range of growth periods, since this type of carbohydrates are known to decrease with increasing growth period (Liljenberg *et al.*, 1995) (McDonald *et al.*, 1991).

Input values on the content of macro and micronutrients and carbon was also needed for the calculation. For 8 of the samples shown in Figure 37, also these elements have been analyzed (Gissén *et al.*, 2014).

Table 26. Grass content of organic matter, carbon and macro and micro nutrients

VS	C	N_{tot}[1]	P	K	Fe	Co	Mo	Ni
		% of DM				mg (kg DM)$^{-1}$		
91	45.3	3.1	0.3	1.8	117	0.8	1.1	1.8

[1] Less than 0.2% of N_{tot} was present as NH_4-N

The VS-value in Table 26 agrees well with the ash content of 9% inFigure 37. Of the total N (N_{tot}) in fresh grass, 75-90% has been described as being bound in proteins (McDonald *et al.*, 1991), and the typical N-content in grass proteins is 16% (Hames *et al.*, 2008). With a protein content of 17% of DM (Table 25) this would mean that 87% of the N_{tot} given in Table 26 would be bound in proteins, which is within the suggested range.

Methane yields experimentally determined by BMP determinations for grass and grass-clover samples from cultivation trials in southern Sweden with growth periods of 48 to 85 days are shown in Figure 39 (Gissén *et al.*, 2014; Prade *et al.*, 2015).The experimental values shown here indicate that the methane yield is inversely correlated to the length of the growth period in the investigated interval. Such a correlation has also been suggested in a previous study on harvest time and number of cuts per year (Prade *et al.*, 2015), and it would be relevant to further investigate the nature of this correlation.

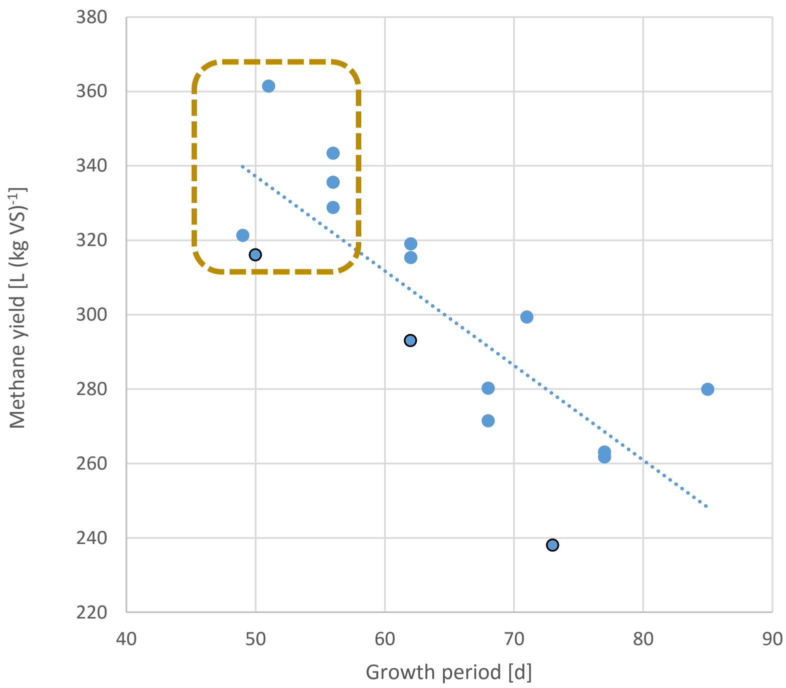


Figure 39. Methane yield in correlation to growth period. For the linear trend line shown in the figure, R^2=0.65. The methane volume is given as dry gas at 0°C and 101 kPa. Data marked with a black circle are own data previously unpublished.

In the present study, however, growth periods in the short range, 42 to 56 days, were used, and the few available experimental data available for that period (encircled in

Figure 39) do not support a decreasing methane yield with increasing growth period. Thus, for the present study, a methane yield that is the same for all investigated harvest dates, within 42-56 days of growth period, is chosen. The value of 334 L (kg VS)$^{-1}$, the average value for the 6 experimental values encircled in Figure 39, is chosen to represent the methane yield that can be achieved in BMP determination. A theoretic methane yield calculated based on the composition in Table 25, with assumed protein and lipid compositions of $C_5H_7O_2N$ and $C_{57}H_{104}O_6$ (Angelidaki and Sanders, 2004) and assuming that "non-structural carbohydrates" is hexose sugars, is 382 L (kg VS)$^{-1}$. Further, the assumption used in the biogas process model is that 5% of the biodegraded VS of the grass is assimilated in microbial biomass, so not available for methane formation. This gives a maximum theoretical methane yield of 363 L (kg VS)$^{-1}$.

The mean value from Table 25 is used as input values on grass composition together with the carbon and nutrient content shown in Table 26 for grass harvested after a growth period in the range of 42-56 days in the present study together with the theoretical methane yield and BMP presented above. These chosen properties for the grass at harvest are used for further calculations in the following chapter.

B.4 BIOCHEMICAL CHANGES DURING WILTING, ENSILING AND FEEDSTOCK OUTTAKE

The data discussed and presented in the previous chapter and summarized in Table 5 are properties for the grass at harvest. In addition, properties will change during wilting, ensiling and outtake of silage. The background data for selecting parameters for different aspects of these losses are presented below. The properties of the grass after losses due to biochemical changes is summarized in Table 5.

B.4.1 Biochemical losses due to aerobic degradation during wilting and the aerobic phase of ensiling

The non-structural carbohydrates most common in temperate grasses are glucose and fructose ($C_6H_{12}O_6$), 1-3% of DM, sucrose ($C_{12}H_{22}O_{11}$), 2-8% of DM, and fructans (polysaccharides), 5-9% of DM (McDonald *et al.*, 1991). Immediately after harvesting, both sucrose and fructans are hydrolysed to glucose and fructose. In the present study, the harvested grass is assumed to be field dried to 35% DM. During wilting, some losses of these sugars will occur. McDonald *et al.* (1991) present a study of wilting of ryegrass-clover with an initial DM content of 17%, where 35% DM was reached already after 6 hours wilting under good conditions. At this stage, no losses of sugars or proteins had occurred. After 48 hours of wilting under god and dry weather conditions, 5% of the sugars and 2% of the proteins where lost, while under moist conditions (100% relative humidity), 1% of sugars and 9% of proteins were lost (McDonald *et al.*, 1991). Proteins in the grass are during a slow, moist, wilt degraded by proteolysis. However, the presence of oxygen during wilting under good, dry, conditions, has been shown to inhibit proteolysis (McDonald *et al.*, 1991). In the present study, wilting and collection is assumed to occur under maximum 48 hours under conditions where oxidation is the dominating process. The calculations are simplified by assuming that no biochemical changes of proteins occur, and that only the carbohydrates are influenced by aerobic degradation, where the sugars are completely oxidized and lost as CO_2.

Other studies have shown that the total DM loss during 2-3 days of wilting of grass under good drying conditions was 3-4% (McDonald *et al.*, 1991). In this case it is not clear if the losses include only biochemical degradation or also mechanical losses. In a

Swedish study, the total loss of DM in field (both mechanical and biochemical losses) was assumed to be 5-8% (Ljungberg *et al.*, 2013).

Also in the initial stage of ensiling, aerobic degradation occurs, since even in a well compacted and sealed silo, oxygen is trapped. This oxygen has been shown to give an aerobic degradation of around 1% of the present hexose-sugar (McDonald *et al.*, 1991).

In the present study, the mechanical losses are accounted for already in the given DM-yields (Table 4) and not included in this section. All biochemical changes are based on degradation of the non-structural carbohydrates, which make up 21% of DM (Table 25) and after harvest will be present as glucose or fructose. 5% of these compounds are assumed to be lost by complete oxidation during wilting and another 1% during the initial, aerobic, phase of ensiling.

Losses in these step will totally under these conditions amount to a DM loss of 1.3% and a methane potential loss of 1.4%.

B.4.2 Biochemical losses due to anaerobic degradation during ensiling

The wilting to 35% DM before ensiling is assumed to give no losses of leachate during ensiling (Liljenberg *et al.*, 1995). During the anaerobic phase of ensiling, the desired reaction is the degradation of easy available organic compounds to lactic acid, which will decrease the pH and inhibit further, unwanted, biochemical degradation. The two main pathways in a well-functioning ensiling process are homolactic and heterolactic fermentation, where the latter also can produce acetic acid, mannitol or ethanol. According to McDonald et al. (McDonald *et al.*, 1991) the DM losses due to fermentation are normally 2-4% in a well-preserved silage, but will give negligible energy losses. Liljenberg et al. (1995) report that a DM loss of around 5% should be expected at ensiling with DM of 35%. Reported DM-losses can however also often be much higher, and one of the reasons can be the difficulty of accurate DM determination for a sample that contains volatile organic acids and alcohols. Losses of these volatiles during oven drying, the standard method of DM determination, will erroneously be reported as a DM loss. Porter & Murray (2001) have shown that 38% of the lactic acid and 98% of the ethanol is lost by evaporation, while Kreuger *et al.* (2011) showed that 100% of the ethanol and 53% of the lactic acid was lost during oven drying of silage. Kreuger *et al.* (2011) have shown that apparent DM-loss for ensiled maize and beets (without correction for the loss of volatile organic compounds during oven drying) is 9 and 38%, while the true DM-loss was 2-3% of DM.

In the present study, 90% of the remaining sugars (glucose and fructose) after the initial aerobic degradation are assumed to be fermented by lactic acid bacteria via heterolactic fermentation according to Eq 1.

[Eq. 1] $C_6H_{12}O_6 – C_3H_6O_3 + C_2H_6O + CO_2$

This will give a DM-loss of 4.4% due to the formation of CO_2, while the methane potential remains unchanged. The calculation is simplified by excluding the fermentation of amino acids, which will also occur.

B.4.3 Biochemical losses due to aerobic deterioration of silage

After opening the silo, or through leakage or poor covering, oxygen access will cause aerobic deterioration of the silage. Lactic acid, acetic acid and other non-structural carbohydrates are the main substrates for the aerobic microorganisms responsible for the aerobic degradation, which are mainly yeasts and moulds. The degraded organic compounds are assumed to be completely oxidized and lost as CO_2.

Silage should be used the same day as it is removed from the silo to minimize losses. Frequent removal also limits the time for air exposure and the aerobic degradation at the face of the silo. It has been shown effective to remove slices of at least 10-30 cm every day to minimize deterioration. This type of biodegradation is strongly temperature dependent, where degradation is much more pronounced during summer. DM-losses for grass silage that has been removed from the silo and exposed to air have been shown to be 7-13% after 7-13 days of air exposure. In another study, DM losses were 0-20% during 7 days of air exposure, with an average of 6.6%.

In the present study, air penetration in the silo, aerobic deterioration at the face of the silo, and air exposure while the silage is taken out and fed into the biogas plant is assumed to cause aerobic degradation and loss of 25% of the non-structural carbohydrates (sugars) and fermentation products (lactic acids and ethanol). These assumptions will give a total DM loss in this stage of 4.1%, which is deemed reasonable in relation to the above figures on 6.6-7% DM loss at 7 days of air exposure. The corresponding loss of methane potential will be 5.4%.

B.5 CULTIVATION INPUTS

For each crop, field operations were modelled in order to calculate energy inputs as consumption of diesel, electricity and heat and other inputs as fertilizer, seeds and pesticides. Indirect inputs like machinery, buildings and infrastructure were included in the economic evaluation, but are not included in the environmental assessment.

B.5.1 Crop specific energy inputs

For each field operation, suitable machinery was selected and corresponding diesel consumption was calculated from typical specific diesel consumption, field capacity and annual use (Maskinkalkylgruppen HIR, 2014). For harvest of grass biomass a yield-dependent model was used to calculate harvest capacities (Prade *et al.*, 2015). On the calculated diesel consumption, another 4% of oil for lubrication was added, and assigned the same emissions as the diesel. In Table 27 to Table 33 energy input (diesel, heat, electricity) is presented for each crop rotation slot in the study.

Table 27. Total energy input [MJ/ha]. **Grass, full production year**

	Scenario Year	C1:m 5	C2:m 5	L:c 2	L:c 3	L:m 2	L:m 3
Operations	Sowing	155	178	164	164	164	164
	Rolling	240	234	229	229	229	229
	Fertilizer spreading	332	290	576	576	576	576
	Grass mowing & windrowing 1	329	367	238	226	316	296
	Grass mowing & windrowing 2	209	204	161	156	191	187
	Grass mowing & windrowing 3	179	195	156	148	187	170
	Grass chopping 1	623	715	546	540	620	578
	Grass chopping 2	533	527	501	498	522	516
	Grass chopping 3	518	530	504	495	525	513
	Transport of 1st harvest	393	354	207	199	258	241
	Transport of 2nd harvest	280	218	146	146	176	168
	Transport of 3rd harvest	257	218	153	138	176	161
	Compaction silo 1	60	50	29	27	39	37
	Compaction silo 2	37	26	16	15	22	20
	Compaction silo 3	30	27	17	15	23	20
	Feed in biogas plant 1	94	77	45	42	61	57
	Feed in biogas plant 2	58	40	25	23	34	31
	Feed in biogas plant 3	48	41	26	22	35	30
Residue spreading	Loading (pumping)	3	3	3	3	3	3
	Transport to fields	132	151	187	187	219	219
	Storage in satellite storage	530	571	541	541	634	634
	Spreading	851	1608	1924	1924	1924	1924
Total	Diesel	4749	5410	5250	5178	5657	5510
	Electricity	0	0	0	0	0	0
	Heat	0	0	0	0	0	0

Table 28. Total energy input [MJ/ha]. **Grass, break year**

	Scenario Year	C1: 6	C2: 6	L:c 4	L:m 4
Operations	Sowing	0	0	0	0
	Rolling	0	0	0	0
	Fertilizer spreading	332	290	576	576
	Grass mowing & windrowing 1	309	315	170	200
	Grass mowing & windrowing 2	174	182	178	209
	Grass mowing & windrowing 3	0	0	0	0
	Grass chopping 1	595	619	504	525
	Grass chopping 2	506	512	510	534
	Grass chopping 3	0	0	0	0
	Transport of 1st harvest	376	307	153	176
	Transport of 2nd harvest	234	191	161	191
	Transport of 3rd harvest	0	0	0	0
	Compaction silo 1	57	42	17	23
	Compaction silo 2	26	21	19	25
	Compaction silo 3	0	0	0	0
	Feed in biogas plant 1	89	67	26	35
	Feed in biogas plant 2	40	33	29	40
	Feed in biogas plant 3	0	0	0	0
Residue spreading	Loading (pumping)	2	1	3	3
	Transport to fields	163	83	511	598
	Storage in satellite storage	322	120	541	634
	Spreading	851	1608	1924	1924
Total	Diesel	3345	3862	4305	4556
	Electricity	0	0	0	0
	Heat	0	0	0	0

Table 29. Total energy input [MJ/ha]. **Winter wheat**

	Scenario Year	C1:c 1	C1:c 4	C1: 1	C1: 4	C2:c 1
Operations	Stubble treatment	231	231	231	231	239
	Ploughing	1060	1060	1060	1060	1133
	Harrowing	264	264	264	264	232
	Sowing	425	425	425	425	496
	Rolling	240	240	240	240	234
	Fertilizer spreading	166	166	166	166	145
	Spraying	241	241	241	241	301
	Combine harvest	948	781	837	781	680
	Transport to farm	319	263	282	263	217
	Drying (ventilator electricity)	1458	1203	1288	1203	894
	Drying (heat production)	3364	2775	2972	2775	2064
	Transport to mill	484	399	428	399	402
Residue spreading	Loading (pumping)	0	0	2	2	0
	Transport to fields	0	0	167	163	0
	Storage in satellite storage	0	0	330	321	0
	Spreading	0	0	851	851	0
Total	Diesel	3985	3694	4653	4552	3667
	Electricity	1458	1203	1288	1203	894
	Heat	3364	2775	2972	2775	2064

Table 30. Total energy input [MJ/ha]. **Winter wheat**

	Scenario Year	C2:c 3	C2:c 4	C2: 1	C2: 3	C2: 4
Operations	Stubble treatment	239	239	239	239	239
	Ploughing	1133	1133	1133	1133	1133
	Harrowing	232	232	232	232	232
	Sowing	496	496	496	496	496
	Rolling	234	234	234	234	234
	Fertilizer spreading	145	145	145	145	145
	Spraying	301	301	301	301	301
	Combine harvest	680	579	635	691	579
	Transport to farm	217	185	203	221	185
	Drying (ventilator electricity)	894	768	840	912	768
	Drying (heat production)	2064	1773	1939	2105	1773
	Transport to mill	402	345	378	410	345
Residue spreading	Loading (pumping)	0	0	2	2	1
	Transport to fields	0	0	239	208	197
	Storage in satellite storage	0	0	349	303	287
	Spreading	0	0	1608	1608	1608
Total	Diesel	3667	3488	5203	5278	5068
	Electricity	894	768	840	912	768
	Heat	2064	1773	1939	2105	1773

Table 31. Total energy input [MJ/ha].

		Spring barley				Oat
	Scenario	**C1:c**	**C1:c**	**C1:**	**C2:c**	**C2:c**
	Year	**3**	**5**	**3**	**5**	**2**
Operations	Stubble treatment	231	231	231	239	239
	Ploughing	1060	1060	1060	1133	1133
	Harrowing	264	264	264	232	232
	Sowing	425	425	425	496	496
	Rolling	240	240	240	234	234
	Fertilizer spreading	166	166	166	145	145
	Spraying	241	241	241	301	301
	Combine harvest	1329	1118	1329	969	479
	Transport to farm	448	377	448	309	153
	Drying (ventilator electricity)	1079	909	1079	342	314
	Drying (heat production)	2490	2097	2490	789	725
	Transport to mill	381	321	381	336	351
Residue spreading	Loading (pumping)	0	0	0	0	0
	Transport to fields	0	0	0	0	0
	Storage in satellite storage	0	0	0	0	0
	Spreading	0	0	0	0	0
Total	Diesel	4354	4033	4354	3957	3370
	Electricity	1079	909	1079	342	314
	Heat	2490	2097	2490	789	725

Table 32. Total energy input [MJ/ha].

		Oat				Oil-seed rape
	Scenario	**C2:c**	**C2:m**	**L:c**	**L:m**	**C1:c**
	Year	**6**	**2**	**1**	**1**	**6**
Operations	Stubble treatment	239	239	367	367	231
	Ploughing	1133	1133	1501	1501	1060
	Harrowing	232	232	310	310	264
	Sowing	496	496	934	934	425
	Rolling	234	234	344	344	240
	Fertilizer spreading	145	145	288	288	166
	Spraying	301	301	498	498	241
	Combine harvest	479	479	390	390	1164
	Transport to farm	153	153	209	209	392
	Drying (ventilator electricity)	314	314	539	539	1046
	Drying (heat production)	725	725	1244	1244	2413
	Transport to mill	351	351	290	290	220
Residue spreading	Loading (pumping)	0	0	0	0	0
	Transport to fields	0	0	0	0	0
	Storage in satellite storage	0	0	0	0	0
	Spreading	0	0	0	0	0
Total	Diesel	3370	3370	4570	4570	3990
	Electricity	314	314	539	539	1046
	Heat	725	725	1244	1244	2413

Table 33. Total energy input [MJ/ha]. Sugarbeet

	Scenario Year	C1:c 2	C1:m 2
Operations	Stubble treatment	231	231
	Ploughing	1060	1060
	Harrowing	264	264
	Sowing	425	425
	Rolling	240	240
	Fertilizer spreading	166	166
	Spraying	561	561
	Inter-row cultivation	357	357
	Harvest of beets	2127	2127
	Field transport	977	977
	Loading to beet cleaner	136	136
	Beet cleaner	108	108
	Transport of dirt	0	0
Residue spreading	Loading (pumping)	0	0
	Transport to fields	0	0
	Storage in satellite storage	0	0
	Spreading	0	0
Total	Diesel	5987	5987
	Electricity	0	0
	Heat	0	0

B.5.2 Material input

The use of seed material is shown in Table 34. The ratio between seed use and assumed seed production is used in the LCA calculations.

Table 34. Seed material use and seed production

	Seeding	Seed production
	[kg ha^{-1}]	
Winter wheat	180	a
Spring barley	170	a
Oats	170	a
Sugar beets	13	700 [b]
Winter oilseed rape	4	a
Grass	7	150 [c]

[a] The actual grain yield in each region (Table 4) is used to calculate the relative impact of seed production in the environmental assessment.
[b] assumption based on multigerm seed rations of 2500-3000 kg ha^{-1}, whereof 20-30% was assumed to be usable after further processing (Kockelmann and Meyer, 2007)
[c] (Strid and Flysjö, 2007)

Buildings included bunker silos for grass biomass ensiling and storage as well as manure pits for storage of slurry and digestate. Emissions for buildings were not included, but bunker silos were assumed to be covered with plastic sheets typical for ensilng purposes and manure pits used for intermediate storage of digestate were assumed to have a plastic roofing to reduce losses of nitrogen, which were included. Plastic and other material inputs in cultivation are summarized in Table 35.

Table 35. Input of active ingredient of pesticide, liming agent, ensiling additive and plastic for ensiling

	Pesticide	Liming agent[b]	Ensiling additive	Plastic[c]
	[kg ha^{-1}]			kg (t DM)$^{-1}$
Winter wheat	1.5-1.6[a]	200	-	-
Spring barley	0.6	200	-	-
Oats	0.6	200	-	-
Sugar beets	4.9	200	-	0.50
Winter oilseed rape	1.2	200	-	-
Grass	-	200	3.7	0.18

[a] Calculated in relation to yield, the lower amount in C2, the higher in C1
[b] Average addition over the crop rotation. Added as 800 kg ha^{-1} limestone every 4th year
c assumptions of heap size and plastic thickness (215 µm thickness, 920 kg m^{-3}) give a need of 1 kg plastic (t DM)$^{-1}$. 50% of the produced beets are assumed to need cover.

Appendix C. Soil carbon modelling

Soil organic carbon (SOC) changes due to changes in cultivation practises, crop rotation or organic amendments are difficult to measure in field experiments, due to a long time frame for the changes and the large inter-field variability. In order to avoid costs for sampling, analyses and error treatment, models have been developed that simulate SOC changes based on the amount and the corresponding quality of organic material added to the soil. One of the models developed in Sweden is ICBM, the Introductory Carbon Balance Model (Andrén and Kätterer, 1997). This model is for instance used for the Swedish national carbon emission balancing required for the Swedish National Inventory report (SEPA, 2015).

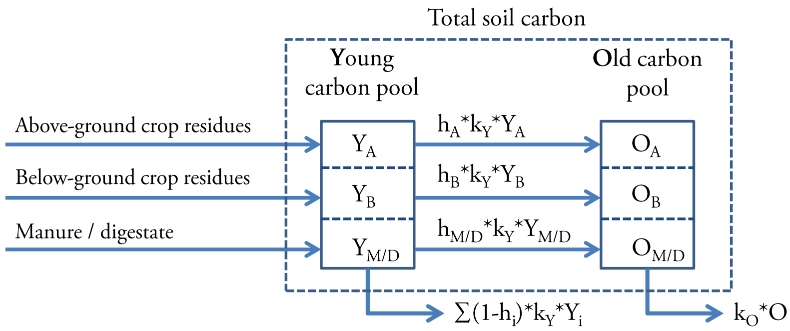


Figure 40. ICBM outline including the different biomass inputs, adjusted from Andrén and Kätterer (Andrén and Kätterer, 1997). Y_i = young carbon pools of different origin; O_i old carbon pools of different origin; h_i humification factors for different residues; k_i reaction coefficients for the different carbon pools.

The ICB model is a two-pool model accounting for two different mineralization rates (Figure 40). Added organic material enters the young carbon pool. A fraction of the organic material described by the humification coefficient (h) continues relatively quickly (50% within less than one year) into the old carbon pool, while the other fraction of carbon is mineralized very quickly and carbon is released to the atmosphere as carbon dioxide. The carbon in the old pool has a much lower mineralization rate (k_O) compared to the young carbon pool and is therefore considered much more stable than that of the young carbon pool. However, without addition of new carbon all carbon of the old carbon pool will be mineralized, i.e. 50% within approx. 100 years.

C.1 AMOUNTS OF CROP RESIDUES

Crop yields – together with the corresponding humification coefficient – play a central role in the simulation SOC development, since these are used to calculate the amount of crop residues potentially added to the soil, where higher crop yields result in larger amounts of crop residues including e.g. straw, stubble, roots and extra root biomass. The model used for calculation of crop residues assumed a linear connection between harvestable biomass (i.e. grains, seeds, beets, above-ground biomass) and remaining residues in the form of fixed mass ratios for the different plant parts and is described in detail by Björnsson *et al.* (2013). Amounts of crop residue relative to the harvested biomass yield are presented in Table 36.
Swedish studies support this model that results in high biomass respective carbon inputs from root and extra root material, especially in grass crops. Grass-legume crops are characterized by a large variability of plant species of grasses and legumes that can

be mixed in endless combinations. While grasses contribute much harvestable biomass, legumes contribute nitrogen fixation and root biomass. In the present study, high-intensity production is assumed which usually results in very low fractions of legume biomass. Another aspect of grass-legume crops is the time factor. High production systems may utilize grass-legume mixtures for 1-3 years.
Root biomass in grass crops is another variable factor. Swedish studies fitting long-term soil carbon measurements to a soil carbon model suggest a constant amount root biomass, 6 t DM/ha (Bertilsson, 2006, 2009). However, in this study, a proportional root biomass development was assumed with a ceiling value of 6 t DM/ha. The Nordic data described above was used to calculate the amounts of crop residues as presented in Table 36.

Straw recovery rates were used in cases where straw was removed from the field (Nilsson and Bernesson, 2009). For sugar beets, a shoot-to-root ratio was used to calculate the amount of above-ground residues. For grass crops, the amount of above-ground residues (stubble) was calculated from the biomass yield and a recovery coefficient (Prade *et al.*, 2015). Below-ground crop residues were calculated in two steps: (a) root biomass and (b) exudates. Root residues were calculated using shoot-to-root ratios, while amounts of exudates (extra-root material) were calculated using an annual extra-root factor of 0.65 (Bolinder *et al.*, 2007). A fraction of 45% C in the dry matter biomass was assumed for all crop residues.

Table 36. Amounts of crop residues (DM) relative to the harvested biomass yield (DM) based on Nordic data (Akhtar and Mashkoor Alam, 1992; Becka *et al.*, 2004; Pietola and Alakukku, 2005; Bolinder *et al.*, 2007; Nilsson and Bernesson, 2009; Arp *et al.*, 2010; Koga *et al.*, 2011; Kätterer *et al.*, 2011).

Crop	Straw/grass		Stubble	Roots	Extra root	Crop residue input	
	field	recovered				Above-ground	Below-ground
Grass crops, establishing year	1.25	1.00	0.00	0.00	0.13	0.00	0.13
Grass crops, full production year	1.25	1.00	0.00	0.00	0.41	0.00	0.40
Grass crops, breaking year	1.25	1.00	0.25	1.48	0.96	0.25	0.75
Oats	0.50	0.32	0.17	0.43	0.28	0.50	0.75 a
Spring barley	0.35	0.18	0.18	0.32	0.21	0.35	0.53
Spring rapeseed	0.90	0.58	0.31	0.31	0.20	0.90	0.52
Sugarbeet	0.30	0.27	0.03	0.01	0.01	0.30	0.02
White mustard	0.67	0.00	0.67	0.51	0.33	0.67	0.83
Winter rapeseed	0.92	0.78	0.14	0.21	0.14	0.92	0.35
Winter wheat	0.57	0.43	0.14	0.31	0.20	0.57	0.52 b

[a] Only oats. In the livestock region, the undersown grass root biomass is included and the value is 0.81-0.84.
[b] Only wheat. In the C1 and C2 regions, the undersown grass root biomass is included and the value is 0.56-0.58.

C.2 ADDITION OF OTHER ORGANIC RESIDUES

In all modified crop rotations, grass biomass is used as biogas substrate. The nutrient-rich biogas residues, digestate, was assumed to be spread as biofertilizer. Besides plant nutrients, organic carbon was therefore added to the soil.

In the current crop rotation of the livestock region, crops were assumed to be cultivated as cattle feed. The resulting manure from the liquid manure handling was assumed to be used as fertilizer on the fields, thereby adding also organic carbon to the soil. In the modified crop rotation of the livestock region, additional grass biomass production was assumed to be used as biogas substrate. Digestate from this biogas production was assumed to be spread on the fields as well.

The amount of carbon added to the soil via manure was calculated from a milk cow manure carbon content of 44% of DM (Rodhe *et al.*, 2013). The amount of carbon added to the soil via digestate was calculated as the amount of carbon removed as methane and carbon dioxide in the biogas process subtracted from the initial amount of carbon in the slurry and the grass biomass, respectively.

C.3 HUMIFICATION COEFFICIENTS

The humification coefficients used in the parameterization of the SOC model were obtained from (Kätterer *et al.*, 2011). A humification coefficient of 0.27 was used for both, undigested and digested cattle slurry as well as for grass digestate. For root biomass, a humification coefficient of 0.35 was used.

Poeplau *et al.* (2015) have shown that the soil clay content impacts the humification coefficient for aboveground crop residues. For soil with high clay content, the humified fraction is higher than in soils with lower clay content. Popleau *et al.* (2015) presented a regression equation for modifying the humification coefficient of litter:

$$h_{aboveground} = -0{,}044 + 0{,}0036 * Clay\ content\ [\%]$$

In order to investigate the impact of soil clay content on the SOC development under the given crop rotations, the mean value of clay content from the European Soil Database Map (Ballabio *et al.*, 2016) was extracted for each study region (C1: 13.9%; C2: 15.6%; L: 5.3%) and the corresponding humification coefficient was calculated according to the equation above. The negative outcome from this equation for the livestock region was set to zero. Due to the low clay content in the livestock region, aboveground residues do not contribute to SOC.

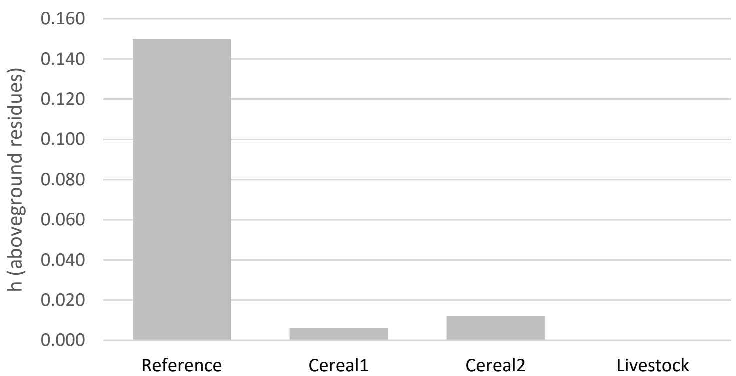


Figure 41. Reference humification coefficient for straw and average humification coefficients for aboveground residues according to average soil clay content in the study regions.

C.4 MODEL CALIBRATION

For each of the cereal regions C1 and C2, the soil carbon model was calibrated against data from a long-term field experiment (

Table 37). For each field experiment, data on annual yields and SOC content determined regularly, was available for two different crop rotations with 16 different fertilization regimes. Calibration was carried out by adjusting the mineralization rate of the old carbon pool (k_0) in order to maximize the coefficient of determination (R^2).

In the livestock region L, no long-term field experiment was available for calibration. Instead, the mineralization rate of the old carbon pool (k_0) was estimated as the mean of the other two regions.

Table 37. Long-term field experiments and initial parameters (KSLA, 2007; Petersen *et al.*, 2008) used in the calibration of the SOC modelling.

Region	Field experiment	Location	Data range	Initial SOC[a] [%]	Clay content [%]	Bulk density [kg/dm³]
C1	Ekebo	55.99° N 12.87° E	1962-2014	3.11	17.80	1.43
C2	Bjertorp	58.24° N 13.13° E	1966-2014	2.18	30.00	1.36

[a] Average value; individual initial SOC values were used for each crop rotation:fertilization regime pair.

C.5 CROP SPECIFIC SOC EFFECTS

Crop-specific SOC effects were calculated as the effect of a specific crop when cultivated continuously. In such a hypothetical cultivation, SOC changes are attributable to the specific crop (in combination with manure or digestate amendments). The annual crop-specific SOC effect was then calculated as the difference in SOC over a period of 40 years divided by 40 years (Björnsson *et al.*, 2013).

Table 38. Carbon input and resulting annual SOC effect [kg/ha/a] for individual crops and scenarios in study region C1.

Year	Crop	Scenario	Crop residues input Above C input	Below C input	Digestat C input	Annual SOC effect
1	Winter	current CR	1764	1603	0	157
		modified CR	1661	1509	0	128
		modified CR + digestate	1661	1509	608	272
2	Sugar beet	current CR	1739	99	0	-277
		modified CR	1739	99	0	-277
		modified CR + digestate	1739	99	0	-277
3	Spring barley	current CR	788	1181	0	21
		modified CR	788	1181	0	21
		modified CR + digestate	788	1181	0	21
4	Winter	current CR	1456	1322	0	71
		modified CR	1456	1417	0	99
		modified CR + digestate	1456	1417	591	238
5	Spring barley	current CR	661	991	0	-36
	Grass	modified CR	0	1755	0	175
	Grass	modified CR + digestate	0	1755	975	405
6	Winter	current CR	1459	563	0	-148
	Grass	modified CR	901	2700	0	460
	Grass	modified CR + digestate	901	2700	592	600

Table 39. Carbon input and resulting annual SOC effect [kg/ha/a] for individual crops and scenarios in study region C2.

Year	Crop	Scenario	Crop residues input			Annual
			Above C input	Below C input	Digestat C input	SOC effect
1	Winter	current CR	1250	1135	0	-183
		modified CR	1199	1089	0	-198
		modified CR + digestate	1199	1089	654	-51
2	Oat	current CR	775	1098	0	-211
		modified CR	775	1098	0	-211
		modified CR + digestate	775	1098	0	-211
3	Winter	current CR	1250	1135	0	-183
		modified CR	1276	1159	0	-175
		modified CR + digestate	1276	1159	568	-48
4	Winter	current CR	1070	972	0	-236
		modified CR	1096	995	0	-229
		modified CR + digestate	1096	995	539	-108
5	Spring barley	current CR	613	919	0	-268
	Grass	modified CR	0	1755	0	-50
	Grass	modified CR + digestate	0	1755	1071	71
6	Oat	current CR	775	1098	0	-211
	Grass	modified CR	1009	2700	0	257
	Grass	modified CR + digestate	1009	2700	226	378

Table 40. Carbon input and resulting annual SOC effect [kg/ha/a] for individual crops and scenarios in study region L.

Year	Crop	Scenario	Crop residues input			Annual SOC effect
			Above C input	Below C input	Digestat C input	
1	Oat	current CR + manure	668	1016	0	-251
		current CR	668	1016	0	-251
		modified CR	668	1041	0	-244
		modified CR + digestate	668	1041	0	-244
2	Grass	current CR + manure	0	1755	804	134
		current CR	0	1755	0	-51
		modified CR	0	1755	0	-51
		modified CR + digestate	0	1755	874	150
3	Grass	current CR + manure	0	1755	804	134
		current CR	0	1755	0	-51
		modified CR	0	1755	0	-51
		modified CR + digestate	0	1755	874	150
4	Grass	current CR + manure	647	2700	804	405
		current CR	647	2700	0	220
		modified CR	871	2700	0	223
		modified CR + digestate	871	2700	874	424

Appendix D. Biogas production in cereal regions

D.1 CHOICE OF PROCESS SIZE AND TYPE

Biogas production in-co-digestion plants is where currently the largest expansion occurs in Sweden (Figure 42). The average national feedstock mix in these plants is shown in Figure 43, and waste from industries/households dominates, corresponding to 800 000 t a^{-1} (SEA, 2015b). It is common with regional competition for the attractive types of waste feedstock for this increasing number of co-digestion plants. In the selected cereal regions, biogas production is occurring today in 4 co-digestion plants in each region (C1 and C2) (Biogasportalen, 2015). Currently, one of these plants is adding small amounts of grass silage[10]. Due to the uncertainty in the availability of and the competition for waste based biogas feedstock with existing biogas plants, the biogas production in the modified scenarios (C1:m, C2:m) were designed based on new construction of biogas plants operating on grass as only feedstock.

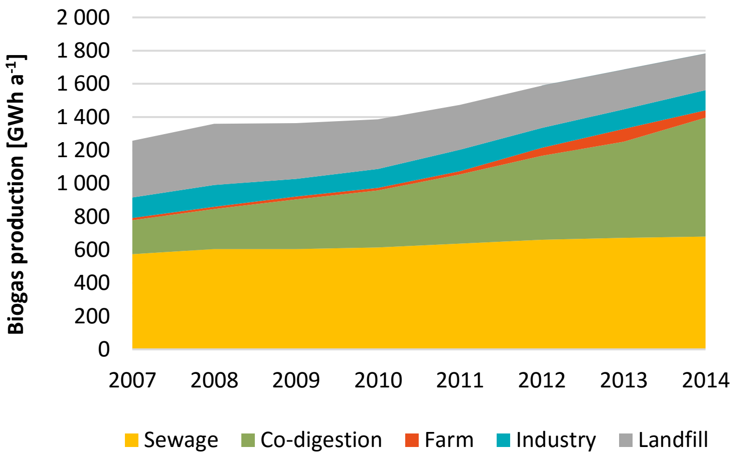


Figure 42. Total annual Swedish biogas production 2007-2014.

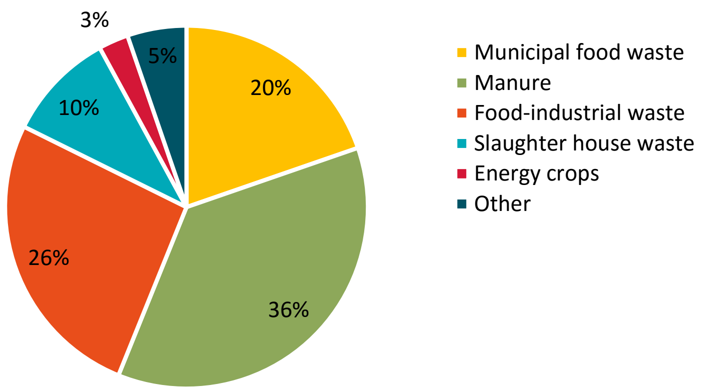


Figure 43. Share of feedstock categories in Swedish co-digestion plants in 2014.

[10] Lars Sjösvärd, Swedish Biogas International, Personal communication June 2015

The Swedish 35 co-digestion plants produced 2.6 PJ of biogas from 1 395 000 t feedstock in 2014 (SEA, 2015b). The average biogas production per plant for these existing co-digestion plants is thus 74 TJ a^{-1}. The majority of the biogas was upgraded, on average 88% of the produced gas during 2010 to 2014 (SEA, 2015b).

The chosen utilization pathway for the biogas in the present investigation is upgrading and use as biofuel for transport. As has been shown by Lantz (2013), upgrading costs per MJ of upgraded gas is highly affected by efficiency of scale up to an installed capacity of approximately 1 000 m^3 biogas per hour. With an assumed methane content of 55% in the biogas, this gives an annual net production of 172 TJ, which was chosen as one plant size for evaluation in the present study. In addition, a plant of half that size, 86 TJ a^{-1}, was evaluated, a size in range with presently operating co-digestion plants.

D.2 PROCESS DESIGN AND CALCULATIONS

The biogas plant was assumed to be a continuous stirred tank reactor (CSTR) operated under mesophilic conditions (37°C), the most common process design for existing Swedish co-digestion plants (SEA, 2015b) and in German biogas production, where crop digestion is more common (FNR, 2010). The reactors were assumed to be ideally stirred, giving effluent concentrations equal to those in the reactor. The calculation model for the biogas production was based on a range of limiting parameters; the maximum organic loading rate (OLR), the minimum hydraulic retention time (HRT) and the maximum DM and total ammoniacal nitrogen (TAN, NH_4^+ and NH_3) concentrations in the reactor (Table 6). The feedstock properties will determine the outcomes, which are the biogas and methane production, the amount of and content in the digestate and the required active reactor volume. In addition, the concentration of the micronutrients Fe, Co, Mo and Ni are calculated and compared to typical requirements. The concentration of TAN is important to consider for high-nitrogen containing feedstocks like grass. Above 3 g l^{-1} TAN in biogas processes has been shown to give a shift towards a slower methanogenic pathway due to the inhibition of aceticlastic methanogenesis (Schnürer and Nordberg, 2008). A longer HRT is thus required to achieve stable operation at high TAN. A survey of 10 Swedish co-digestion plants showed that the average concentration of TAN was 3.0 g l^{-1}, with a maximum value of 6.7 g l^{-1}, showing that conventional biogas plants can be operated within that range of concentrations (Ljung *et al.*, 2013). Experimental trials have shown that stable operation can be achieved at 5.3 g l^{-1} TAN with 56 d HRT (Schnürer and Nordberg, 2008). The maximum limit for TAN was here set at 5.0 g l^{-1}, in combination with a minimum HRT of 50 d. The chosen limits for all process parameters are shown in Table 6. The micronutrient level is also important for a well-functioning process. Minimum concentrations for micronutrients as chosen by Lantz *et al.* (2013) were 50, 0.5, 0.5 and 0.2 g l-1 for Fe, Co, Mo and Ni respectively.

The calculation model used was the same as presented by Lantz *et al.* (2013), where the methane yield for the feedstock and the methane content in the biogas are determining the conversion of organic matter. The calculations are based on the assumption that organic matter (VS) converted to CH_4 or CO_2 is removed from the feedstock content. No other compounds are assumed to be lost through the biogas, e.g. water, hydrogen sulphide or ammonia. The organic matter removed through the biogas was assumed to represent 95% of the total metabolized mass of C, H and O with the remaining 5% assimilated into microbial biomass (McCarty, 1964).

The fraction of organically bound nitrogen converted to TAN was calculated by assuming that the degree of mineralization was equal to the degree of VS metabolization (Lantz *et al.*, 2013). The TAN was then reduced by the amount of N integrated in new biomass by microbial assimilation. The amount of TAN assimilated was set to 11% (w/w) of the sum of C, H, O and N in the microbial biomass produced, (McCarty, 1964).

D.3 PROCESS OUTCOMES

The calculated outcomes for the two sizes of biogas plant are summarized in Table 7. The TAN concentration in the reactor became a limiting factor, and in the calculations, water was added for dilution. Due to this dilution, the HRT decreased to 42 days, and the OLR had to be decreased to reach the desired minimum of 50 d HRT. The actual process design parameters are shown in Table 6 together with the set limits. The outcomes as reactor tank size, feedstock quantities needed and gas flows are shown in Table 45. The processes for regions C1 and C2 are identical, since no difference is made in grass properties for these two regions.

The amounts of the digestate are shown in Table 8 together with the composition before and after storage loss, which will be the same irrespective of plant size. The losses during storage are further explained in Chapter 0.

The concentrations of micronutrients based on feedstock content (Table 26), mass losses and water dilution in the process was calculated to 29, 0.3, 0.2 and 0.5 mg kg^{-1}, for Fe, Co, Mo and Ni respectively. These concentrations are lower, but in a similar range as the limits (50, 0.5, 0.5 and 0.2) set by (Lantz *et al.*, 2013)). Those limits are, however, presented as being in the high range. A literature review of data on recommended and actually present concentrations of micronutrients in biogas processes have also been shown to vary much, by 1-2 orders of magnitude (Schattauer *et al.*, 2011). The micronutrients present in the grass are thus considered to be sufficient for a well-functioning process.

D.4 FEEDSTOCK STORAGE AND HANDLING

The handling of the grass to be used for biogas production is presented in Chapter 6.3.2. The field drying to 35% DM means that water for dilution later has to be added in the process, and a DM of 20% in the grass at feed-in would have given perfect conditions in the digester without need for dilution. However, field drying is necessary to minimize losses at ensiling, and the operators using grass for biogas production today are applying the same principle, with a DM of 33-35% at feed-in[11]. The losses during ensiling and handling and the biochemical background is presented in Appendix B.

D.5 DIGESTATE STORAGE AND HANDLING

The digestate was assumed to be stored in covered storage tanks, with storage capacity sufficient for 12 months of digestate production. The reason for the somewhat over dimensioned storage capacity was that it was desired to focus all digestate application as biofertilizer in the spring. This will minimize the amount of digestate in storage

[11] Gunnar Hagsköld, Växtkraft, & Lars Sjösvärd, Swedish Biogas International. Personal communication June 2015.

during the warmer period of the year, which will reduce the risk for methane leakage from storage, and avoid biofertilizer application in autumn, which will reduce the risk for nitrogen leakage and nitrous oxide emissions. The impact of this strategy on the share of digestate in storage the warmer months is illustrated in Figure 44.

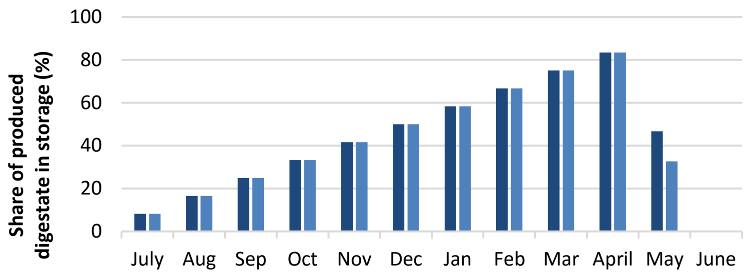


Figure 44. The share of the total produced digestate that is kept in storage each month for region C1 (dark blue) and C2 (light blue).

Calculation of losses of methane during storage are based on the amount of organic matter (given as VS) from the digester and the maximum methane potential of this organic matter (B_0), based on the model proposed by IPCC (2006). The B_0 of the grass (379 m^3 (t VS)$^{-1}$, Table 5) is then used as the starting point, and with the used methane yield in the process of 281 m^3 (t DM)$^{-1}$ the B_0 of the digestate is calculated to 238 m^3 (t VS)$^{-1}$. The methane conversion factor (MCF) describes the share of this theoretical methane yield that will be actually produced during storage. The MCF for liquid manure storage of 3.5% used in the latest Swedish national inventory report (SEPA, 2015) is here applied also for the digestate. Very high MCF-values (29%) has been shown for digestate storage during summer (Rodhe *et al.*, 2013), but with biofertilizer application in spring/early summer, the share of digestate kept in storage during the warmer months is low (Figure 44), and MCF-values for winter storage has been shown to be only 0.1% (Rodhe *et al.*, 2013), which motivates the use of an annual average MCF of 3.5%, even if summer emissions would be as high as the above study has shown.

For ammonia losses during digestate storage, the same conditions as for liquid manure storage with no floating crust and under roof cover are assumed (SEPA, 2015) giving a loss of NH_3-N corresponding to 1% of N-tot, and with no N_2O emissions.

D.6 BIOGAS UPGRADING

The biogas produced is assumed to be utilized as vehicle fuel. To fulfil the Swedish standard for biogas utilized as vehicle fuel various contaminants must be removed and the methane content must be increased to 95 – 99 %. This upgrading could be achieved by using different technologies such as water scrubber, amine scrubber and membrane (Bauer *et al.*, 2013).

In this study calculations are based on the chemical scrubber due to its low electricity requirements and low methane losses.

After upgrading, the biogas is distributed to various filling stations. In Sweden, approximately 30% is distributed via the natural gas grid or the vehicle gas grid in Stockholm. The remaining gas is distributed via local grids or by truck (SEA, 2015b).

Here, it is assumed that the biogas is compressed at site from 0.4 to 20 MPa and then distributed by truck.

D.7 ENERGY INPUT AND EMISSIONS

Different biogas plants have different energy requirements depending on feedstock, scale and process design. In this study, two plants with the same feedstock composition but in different scale are evaluated. Since they are both in industrial scale it is assumed that energy requirements are the same. The energy inputs and emissions described below are summarized in Table 9.

As presented earlier it is assumed that the biogas plant is operated under mesophilic conditions (37^{o}C). Assuming a feedstock temperature of 8 ^{o}C and a heat capacity for DM of 1 MJ (t K)$^{-1}$, the theoretical heat demand is calculated to 119 MJ/t including 15 % heat losses. Additional heat exchange between digestate and feedstock could be possible but is not included here.

Based on the literature review presented by Lantz *et al.* (2013) electricity demand is set to 29 MJ/t feedstock.

Energy requirements for the upgrading plant is set to 0.12 kWh electricity and 2.2 MJ heat per m^3 of biogas. It is also assumed that waste heat from the upgrading process could be utilized to heat the biogas process. This would mean that 63% of the heat required for upgrading needs to be available for heating the biogas process, which is possible since up to 80% of the heat can potentially be made available[12].

Regarding methane losses it is assumed that the biogas plant and the upgrading plant is constructed according to best available technology resulting in 0.5 % and 0.1 % methane losses respectively (Lantz *et al.*, 2013; Tufvesson *et al.*, 2013).

D.8 TRANSPORT DISTANCE

The transport distance for the grass to the biogas plant is calculated based on the land use illustrated in Figure 5 and Figure 6 (Table 41).

Table 41. Land use in the selected regions

	Arable land current crop rotation	Arable land other crops	Non-arable land
C1 [ha]	196687	155148	227567
C2 [ha]	77280	160589	362931

In the modified crop rotations, grass is cultivated two out of six years in the crop rotations in both regions, which with the grass yields presented in Table 20, ensiling/handling losses subtracted, and the methane yield as in Table 5 , the methane production corresponds to 24 and 27 GJ ha^{-1} respectively for regions C1 and C2 (as average for the whole crop rotation, the methane yield per ha of grass is 73 (C1) and 81 (C2) GJ ha^{-1}). The transport distance is calculated based on the following assumptions:

- The average use of arable land surrounding the biogas plant is distributed as for the regions in general (Table 41)

[12]Lars-Evert Karlsson, Purac Puregas, Personal communication

- The grass crop rotations are implemented in all current crop rotations surrounding the biogas plant.
- The grass is collected from arable land surrounding the biogas plant with a circular geometry.

The average transport distance from the field to the biogas plant (*T*) is then calculated using Eq. 2, where *r* is the radius of the land area needed (total land area, including non-arable land) and τ is the tortuosity factor (Overend, 1982). The tortuosity factor indicates the relation between the road distance and a straight line, and is assumed to be 1.3 for both regions (Börjesson and Gustavsson, 1996).

$$T = 2/3\ r\ \tau \qquad \text{[Eq. 2]}$$

The relation between biogas production per plant and the required transport distance for the two regions is shown in Figure 45. For the 86 TJ a^{-1} biogas plants, the distance will be 5.0 and 7.7 km for C1 and C2 respectively, and for the 172 TJ a^{-1} plants 7.1 and 10.9 km. The same transport distance is used for transports of digestate to satellite storage tanks.

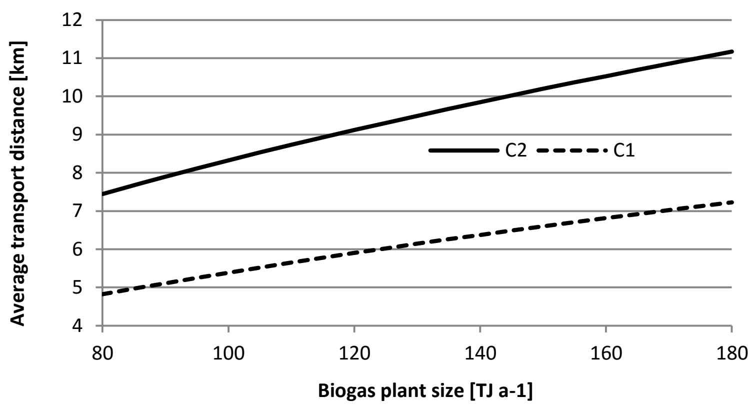


Figure 45. The average one-way transport distance between field and biogas plant depending on biogas plant size for the two regions. The biogas plant size is given as produced methane before losses.

Appendix E. Biogas production in the livestock region

Grass in the livestock regions current crop rotation is used as cattle feed. The typical feed demand for milking cows and the production in the livestock region is shown in Table 42. Based on the crop rotation and yields assumed at current conditions, one ha of arable land in the livestock region can supply 0.5-1.2 milk cows with feed. The average density of cattle in the region in 2013 was 0.69 animal units (AU) ha^{-1} and other categories of livestock were very low (Figure 36). The calculations for cultivation conditions, manure amounts and grass availability were based on an assumed cattle density of 0.7 AU ha^{-1}. The manure presently used as feedstock at the biogas plant (55 055 t a^{-1}) would then originate from an area of arable land corresponding to 3 000 ha (Table 42). The grass cultivated here under current conditions was assumed to be completely used as cattle feed, and the cattle manure (Table 42) to be used as biofertilizer in the cultivation of this grass.

In the modified scenario, the same crop rotation is maintained, but grass cultivation is intensified, with a yield increase as shown in Table 4. This additional grass is assumed to be used as biogas feedstock in addition to the manure, replacing the waste (Table 10). The digestate resulting from manure and grass digestion in the modified scenario was replacing undigested manure as biofertilizer in grass cultivation. The conditions in biogas production in the reference biogas plant in 2014, and in the modified scenario are shown in Table 11.

Table 42. Calculation of potential animal and resulting slurry production

Parameter	Unit	Value	Reference
Milk production	[kg/AU/a]	8998	(Strid *et al.*, 2012)
Feed requirements			
Silage	[kg/AU/a]	3367	(Strid *et al.*, 2012)
Cereals	[kgAU//a]	1620	(Strid *et al.*, 2012)
Crop production[1]			
Grass-clover crops	[kg/ha/a]	4875	
Oat	[kg/ha/a]	749	
Storage losses & rejections	[%]	20	(Strid *et al.*, 2012)
Feed production			
Silage	[kg/ha/a]	3900	own calculation
Cereals	[kg/ha/a]	749	own calculation
Potential animal production			
Silage	[AU²/ha]	1.2	own calculation
Cereals	[AU²/ha]	0.5	own calculation
Current cattle density[3]	[AU²/ha]	0.7	(SJV, 2008)
Slurry production			
Slurry amount	[m^3/AU/a]	26.2	(SFS, 2013)
Slurry amount	[m^3/ha/a]	18.3	own calculation

[1] Corresponding to yields in the current livestock scenario.
[2] AU = animal units; 1 milk cow = 1 animal unit. Current cattle density is shown in Figure 36.
[3] Lacking cereal is assumed to be purchased and excess silage is assumed to be sold

With grass and manure as biogas feedstocks in the modified scenario, the total methane production will increase by 40% to totally 63 TJ a^{-1}. The biogas upgrading capacity at the plant is dimensioned for 720 m^3 biogas per hour, so from that aspect, the plant can handle a higher production even if the compressor capacity presently is lower[13]. A notable feature for the modified process is, however, the high level of TAN, 4.0 g l^{-1}. Biogas processes have been shown to operate well at high levels of ammonia present, but a shift to a slower degradation pathway is occurring, and a high TAN should be accompanied with a high hydraulic retention time to get efficient degradation and to not risk overload/acidification of the process (Schnürer and Nordberg, 2008). In a previous study, the TAN limit was set to 4.0 g l^{-1} in combination with a HRT of at least 46 days based on observations from German biogas plants operating with a feedstock base of manure and energy crops (FNR, 2010; Lantz *et al.*, 2013). Here, the HRT is at 42 days, which could be on the low side. The DM concentration in the modified process is assumed to be manageable, although in an existing process designed for lower DM, modifications could be needed.

E.1 FEEDSTOCK STORAGE AND HANDLING

Grass silage is transported to the biogas plant and stored in a bunker silo. Since the silage used for biogas production is cut to 4 mm it might be possible to use existing feed in equipment. However, in order to minimize the risk for process disturbances, it is assumed that silage is feed into the biogas reactor with a separate system including a hammer mill or similar. Based on discussion with market actors the investment cost for such system is estimated to 200 000 €[14].

E.2 DIGESTATE STORAGE AND HANDLING

The amount of digestate in the L:m scenario is almost the same as in the L:c scenario. It is therefore assumed that no additional digestate storages are required.

E.3 ENERGY INPUT AND EMISSIONS

The energy input and emissions for biogas production in the livestock region are based on current operational conditions at the existing biogas plant, and are summarized in Table 12.

E.4 TRANSPORT DISTANCE

The transport distance for the grass to the biogas plant is calculated based on the share of arable land in the region, and based on the modified scenario for crop cultivation assuming that ¾ of the arable land is used for grass cultivation. The cattle manure used in the biogas plant is assumed to originate from 2 100 AU (Table 42), which with the cattle density in the region (0.7 AU ha^{-1}) are supported by 3 000 ha arable land. The intensified grass production assumed in the modified scenario, providing on average an extra 2.25 t DM ha^{-1} for biogas production, occurs on ¾ of this arable land. The grass cultivation areas (4.2% of the total land area) are assumed to be evenly distributed over the total land area in the region (Table 43). The transport distance is calculated based on the same formula and assumptions about geometry and distribution as presented for the cereal regions and will be 11.4 km one way from the field to the biogas plant. The

[13] Björn Goffeng, Göteborg Energi. Personal communication

[14] Björn Goffeng, Göteborg Energi and Olof Petersson, Purac, personal communication.

transport distance for the distribution of the digestate will be the same (as it is used as biofertilizer in grass cultivation).

Table 43. Land use in the livestock region (ha).

Non-arable land	Arable land	Whereof arable land for Grass cultivation
1 258 153	74 044	55 533 [a]

[a] Based on the assumed modified crop rotation with 3 years grass and one year oats. The actual area for cultivation of grass was on average 2010-2014 59 900 ha (Olsson, 2015).

Appendix F. Energy prices

In this study, the calculated production cost for crops as well as biogas includes the cost for electricity and wood chips based on Swedish market prices. The economic feasibility of analyzed biogas systems is also highly effected by estimated market price for the biogas produced. Assumptions made for each energy carrier are presented below.

F.1 ELECTRICITY

Market price for electricity vary considerable over time. In this study the cost of electricity is set to 38 €/MWh which is the average price for 1 year contracts signed in 2015 by companies within the agricultural sector (SCB, 2016b). In addition to the energy price the transmission fee is set to 36 €/MWh for approximately the same period of time and the tax is set to 0.5 €/MWh (SCB, 2016b; Skatteverket, 2016). In total, the cost for electricity used within the agricultural sector is set to 74 €/MWh.

Since biogas plants probably use more energy than agricultural companies the cost for electricity is here set to 69 €/MWh.

F.2 HEAT

In 2015, the average price for wood chips used by producers of district heating was approximately 5.5 €/GJ (SEA, 2016). In Lantz and Börjesson (2014), the total cost for heat from wood chips are assumed to be 15 €/GJ heat including cost of capital as well as operation and maintenance. This is also used in this study. For comparison, the average market price for district heating sold to multi-dwelling buildings in Sweden are 18 €/GJ (SEA, 2016).

F.3 BIOGAS

In this study, it is assumed that biogas is utilized as vehicle fuel in busses, passenger cars and trucks. In 2015 there was 211 filling stations for vehicle gas in Sweden. More than 70 % of these filling stations was available for the public and the rest are used by bussed and other dedicated vehicle fleets. More than 50 % of the vehicle gas was however used by busses (SCB, 2016a).

On public filling stations the average market price in April 2016 was 1.4 €/kg + VAT (Gasbilen, 2016). Given an energy density of 13 kWh/kg this corresponds to 135 €/MWh or 30.4 €/GJ + VAT. At individual stations, the price varies from 1.1 to 1.6 €/kg + VAT.

For filling stations that are not public there are no official price although it seems likely that it is lower due to efficiency of scale since non-public filling stations in general handle more gas than public stations.

The market price presented here includes production of vehicle gas, distribution (including transportation and filling stations) and profit margin for all actors involved in the biogas system. The distribution cost depends on how the gas in transported (gas grid or by truck), transportation distance and how much gas that is sold on each filling station. Thus, this part of the biogas system could vary significantly depending on local conditions for each biogas system. In Hagberg *et al.* (2016), distribution cost is set to 7,5

€/GJ which could be compared to Dahlgren *et al.* (2013), who estimated the distribution cost to 10.3 – 14.8 €/GJ depending on transportation distance. The lower number represent "normal" transportation distance.

In this study it is assumed that the biogas producer could sell upgraded and compressed biogas at the biogas plant for 21.5 €/GJ assuming a distribution cost of 8.9 €/GJ. The effect of a higher or lower gas price are also evaluated in the sensitivity analysis.

Appendix G. Crop production costs

Crop production costs were estimated using the same production systems as assumed in the calculation of the cultivation energy inputs (0) including relevant operations as described earlier (8.1).
Costs were calculated in Swedish Crowns (SEK) and converted to Euro (EUR, €) using an exchange rate of 9.4 SEK/€.

G.1 PRODUCTION MEANS

Costs of energy, fertilizer and ensiling plastics used in the production of food, feed and energy crops were assumed representative of the 2014 price index (Table 44).

Table 44. Specific costs of selected materials.

Material	Unit	Costs
Electricity	[€-ct./kWh]	7.45
Fertilizer N	[€/kg]	0.96
Fertilizer P	[€/kg]	2.13
Fertilizer K	[€/kg]	0.85
Ensiling plastics	[€/m²]	2.98

G.2 MACHINERY COSTS

Machinery costs were calculated from annual use [h/ha] as estimated for the diesel consumption calculations (0) and corresponding hourly costs, including costs for fuel use and driver (Maskinkalkylgruppen HIR, 2014).

G.3 BUILDINGS

Buildings used in the calculations included ensiling plastic-covered bunker silos for biogas substrate storage, cereal tower silos and covered digestate wells. Costs for bunker silos and cereal tower silos were calculated as annualized investment costs per m³ storage volume

Table 45). Costs for the digestate well (Maskinkalkylgruppen HIR, 2014) were complemented with a plastic cover assuming a 1.5 times costs increase.

Table 45. Specific costs of storage facilities and annual volume requirements as part of total volume of substrate, cereals and digestate, respectively.

Storage type	Costs [€/m³/a]	Annual volume coverage [%]
Substrate bunker silo excl. ensiling plastics	1.55	100
Cereal tower silo	3.70	100
Digestate well, incl. cover	2.88	90

G.4 CROP-SPECIFIC ECONOMIC COSTS

In Table 46 to Table 52 economic costs are presented for each crop rotation slot in the study.

Table 46. Economic costs [€/ha].

		Grass, full production year					
	Scenario	**C1:m**	**C2:m**	**L:c**	**L:c**	**L:m**	**L:m**
	Year	**5**	**5**	**2**	**3**	**2**	**3**
Material	Fertilizer N	150	165	155	144	149	134
	Fertilizer P	3	5	-3	-6	-9	-12
	Fertilizer K	86	99	44	34	54	40
	Seeds	31	31	31	31	31	31
	Pesticides	9	9	9	9	9	9
	Liming	5	5	5	5	5	5
Operations	Sowing	18	18	16	16	16	16
	Rolling	24	25	22	22	22	22
	Fertilizer spreading	35	32	56	56	56	56
	Grass mowing & windrowing 1	31	24	14	14	18	17
	Grass mowing & windrowing 2	23	16	14	14	14	14
	Grass mowing & windrowing 3	23	16	14	14	14	14
	Grass chopping 1	33	30	19	19	19	19
	Grass chopping 2	31	24	19	19	19	19
	Grass chopping 3	31	24	19	19	19	19
	Transport of 1st harvest	58	49	24	23	26	26
	Transport of 2nd harvest	44	32	21	21	23	21
	Transport of 3rd harvest	41	32	21	21	23	21
	Compaction silo 1	6	6	3	3	4	4
	Compaction silo 2	4	3	2	2	2	2
	Compaction silo 3	3	3	2	2	2	2
	Feed in biogas plant 1	9	9	5	5	7	6
	Feed in biogas plant 2	6	5	3	3	4	3
	Feed in biogas plant 3	5	5	3	2	4	3
Storage	Concrete bunker silo	57	65	42	39	57	53
	Plastic cover for ensiling	26	29	19	18	26	24
Residue spreading	Loading (pumping)	0	0	0	0	0	0
	Transport to fields	49	55	59	59	69	69
	Storage in satellite storage	76	80	70	70	82	82
	Spreading	106	200	235	235	235	235
Production steps	Cultivation	467	589	569	546	568	536
	Harvest	171	134	97	97	102	101
	Storage	172	185	139	133	174	167
	Transport	212	186	136	133	155	150
	Total	**1021**	**1094**	**941**	**909**	**998**	**954**
Production means	Machinery	472	485	456	454	474	470
	Diesel	106	122	113	112	121	118
	Electricity	0	0	0	0	0	0
	Heat	0	0	0	0	0	0
	Liming	5	5	5	5	5	5
	Seeding material	31	31	31	31	31	31
	Pesticides	9	9	9	9	9	9
	Plastic	26	29	19	18	26	24
	Buildings	133	144	113	109	139	135
	Fertilizer	240	269	196	172	194	162
	Total	**1021**	**1094**	**941**	**909**	**998**	**954**

Table 47. Economic costs [€/ha]. **Grass, break year**

		C1:	C2:	L:c	L:m
	Scenario Year	**6**	**6**	**4**	**4**
Material	Fertilizer N	134	169	97	70
	Fertilizer P	4	22	-18	-28
	Fertilizer K	60	101	-10	-19
	Seeds	31	31	31	31
	Pesticides	9	9	9	9
	Liming	5	5	5	5
Operations	Sowing	0	0	0	0
	Rolling	0	0	0	0
	Fertilizer spreading	35	32	56	56
	Grass mowing & windrowing 1	29	21	14	14
	Grass mowing & windrowing 2	23	16	14	14
	Grass mowing & windrowing 3	0	0	0	0
	Grass chopping 1	32	26	19	19
	Grass chopping 2	31	24	19	19
	Grass chopping 3	0	0	0	0
	Transport of 1st harvest	55	42	21	29
	Transport of 2nd harvest	39	30	21	31
	Transport of 3rd harvest	0	0	0	0
	Compaction silo 1	6	5	2	2
	Compaction silo 2	3	2	2	3
	Compaction silo 3	0	0	0	0
	Feed in biogas plant 1	9	8	3	4
	Feed in biogas plant 2	4	4	3	4
	Feed in biogas plant 3	0	0	0	0
Storag	Concrete bunker silo	37	40	24	33
	Plastic cover for ensiling	17	18	11	15
Residue :	Loading (pumping)	0	0	0	0
	Transport to fields	41	18	101	118
	Storage in satellite storage	46	17	70	82
	Spreading	106	200	235	235
Production steps	Cultivation	384	569	405	359
	Harvest	114	87	65	65
	Storage	109	83	110	135
	Transport	147	102	149	186
	Total	**754**	**841**	**728**	**745**
Production means	Machinery	336	341	414	443
	Diesel	75	88	95	104
	Electricity	0	0	0	0
	Heat	0	0	0	0
	Liming	5	5	5	5
	Seeding material	31	31	31	31
	Pesticides	9	9	9	9
	Plastic	17	18	11	15
	Buildings	83	57	95	115
	Fertilizer	199	293	69	23
	Total	**754**	**841**	**728**	**745**

Table 48. Economic costs [€/ha].		Winter wheat				
	Scenario	**C1:c**	**C1:c**	**C1:**	**C1:**	**C2:c**
	Year	**1**	**4**	**1**	**4**	**1**
Material	Fertilizer N	114	131	59	68	129
	Fertilizer P	53	43	20	18	38
	Fertilizer K	29	24	-29	-29	21
	Seeds	84	84	84	84	84
	Pesticides	111	111	111	111	94
	Liming	5	5	5	5	5
Operations	Stubble treatment (cultivator)	25	25	25	25	29
	Ploughing	108	108	108	108	116
	Harrowing	27	27	27	27	25
	Sowing	56	56	56	56	82
	Rolling	24	24	24	24	25
	Fertilizer spreading	17	17	17	17	16
	Spraying	33	33	33	33	41
	Combine harvest	103	85	91	85	77
	Transport to farm	25	21	22	21	17
	Drying (ventilator electricity)	101	84	90	84	67
	Drying (heat production)	135	112	120	112	89
	Transport to mill	47	38	41	38	39
Storage	Cereal Silo	0	0	0	0	0
Residue spreading	Loading (pumping)	0	0	0	0	0
	Transport to fields	0	0	42	41	0
	Storage in satellite storage	0	0	47	46	0
	Spreading	0	0	106	106	0
Production steps	Cultivation	687	689	647	654	704
	Harvest	103	85	91	85	77
	Storage	275	227	290	273	184
	Transport	72	59	105	100	55
	Total	**113**	**106**	**1134**	**1111**	**102**
Production means	Machinery	373	349	485	476	380
	Diesel	93	86	108	106	85
	Electricity	101	84	90	84	67
	Heat	135	112	120	112	89
	Liming	5	5	5	5	5
	Seeding material	84	84	84	84	84
	Pesticides	111	111	111	111	94
	Plastic	0	0	0	0	0
	Buildings	38	32	81	78	28
	Fertilizer	196	199	51	57	188
	Total	**113**	**106**	**1134**	**1111**	**102**

Table 49. Economic costs [€/ha].

		Winter wheat				
	Scenario	**C2:c**	**C2:**	**C2:**	**C2:**	**C2:**
	Year	**3**	**4**	**1**	**3**	**4**
Material	Fertilizer N	129	116	39	70	58
	Fertilizer P	38	33	8	14	10
	Fertilizer K	21	18	-39	-29	-30
	Seeds	84	84	84	84	84
	Pesticides	94	94	94	94	94
	Liming	5	5	5	5	5
Operations	Stubble treatment (cultivator)	29	29	29	29	29
	Ploughing	116	116	116	116	116
	Harrowing	25	25	25	25	25
	Sowing	82	82	82	82	82
	Rolling	25	25	25	25	25
	Fertilizer spreading	16	16	16	16	16
	Spraying	41	41	41	41	41
	Combine harvest	77	66	72	78	66
	Transport to farm	17	14	16	17	14
	Drying (ventilator electricity)	67	58	63	68	58
	Drying (heat production)	89	77	84	91	77
	Transport to mill	39	33	36	39	33
Storage	Cereal Silo	0	0	0	0	0
Residue spreading	Loading (pumping)	0	0	0	0	0
	Transport to fields	0	0	53	46	43
	Storage in satellite storage	0	0	49	42	40
	Spreading	0	0	200	200	200
Production steps	Cultivation	704	683	724	771	754
	Harvest	77	66	72	78	66
	Storage	184	158	222	230	198
	Transport	55	47	105	102	91
	Total	**102**	**954**	**1123**	**1182**	**1109**
Production means	Machinery	380	365	589	591	572
	Diesel	85	81	121	123	118
	Electricity	67	58	63	68	58
	Heat	89	77	84	91	77
	Liming	5	5	5	5	5
	Seeding material	84	84	84	84	84
	Pesticides	94	94	94	94	94
	Plastic	0	0	0	0	0
	Buildings	28	24	75	71	64
	Fertilizer	188	167	8	55	38
	Total	**102**	**954**	**1123**	**1182**	**1109**

Table 50. Economic costs [€/ha].		**Spring barley**				**Oat**
	Scenario	**C1:**	**C1:**	**C1:**	**C2:**	**C2:**
	Year	**3**	**5**	**3**	**5**	**2**
Material	Fertilizer N	80	86	80	85	80
	Fertilizer P	43	36	43	33	30
	Fertilizer K	22	18	22	17	15
	Seeds	70	70	70	70	70
	Pesticides	53	53	53	33	33
	Liming	5	5	5	5	5
Operations	Stubble treatment (cultivator)	25	25	25	29	29
	Ploughing	108	108	108	116	116
	Harrowing	27	27	27	25	25
	Sowing	56	56	56	82	82
	Rolling	24	24	24	25	25
	Fertilizer spreading	17	17	17	16	16
	Spraying	33	33	33	41	41
	Combine harvest	144	121	144	110	54
	Transport to farm	35	30	35	24	12
	Drying (ventilator electricity)	75	63	75	41	37
	Drying (heat production)	100	84	100	54	50
	Transport to mill	37	31	37	32	34
Storage	Cereal Silo	0	0	0	0	0
Residue spreading	Loading (pumping)	0	0	0	0	0
	Transport to fields	0	0	0	0	0
	Storage in satellite storage	0	0	0	0	0
	Spreading	0	0	0	0	0
Production steps	Cultivation	562	558	562	576	567
	Harvest	144	121	144	110	54
	Storage	208	175	208	120	118
	Transport	72	60	72	56	45
	Total	**986**	**914**	**986**	**862**	**785**
Production means	Machinery	405	379	405	407	355
	Diesel	101	94	101	92	78
	Electricity	75	63	75	41	37
	Heat	100	84	100	54	50
	Liming	5	5	5	5	5
	Seeding material	70	70	70	70	70
	Pesticides	53	53	53	33	33
	Plastic	0	0	0	0	0
	Buildings	33	27	33	25	31
	Fertilizer	144	140	144	135	125
	Total	**986**	**914**	**986**	**862**	**785**

Table 51. Economic costs [€/ha].

		Oat				Oil-seed rape
	Scenario	**C2:c**	**C2:m**	**L:c**	**L:m**	**C1:c**
	Year	**6**	**2**	**1**	**1**	**6**
Material	Fertilizer N	80	80	64	64	136
	Fertilizer P	30	30	25	25	49
	Fertilizer K	15	15	13	13	26
	Seeds	70	70	70	70	43
	Pesticides	33	33	33	33	149
	Liming	5	5	5	5	5
Operations	Stubble treatment (cultivator)	29	29	36	36	25
	Ploughing	116	116	149	149	108
	Harrowing	25	25	30	30	27
	Sowing	82	82	104	104	56
	Rolling	25	25	33	33	24
	Fertilizer spreading	16	16	28	28	17
	Spraying	41	41	62	62	33
	Combine harvest	54	54	48	48	126
	Transport to farm	12	12	13	13	31
	Drying (ventilator electricity)	37	37	40	40	47
	Drying (heat production)	50	50	54	54	63
	Transport to mill	34	34	28	28	21
Storag	Cereal Silo	0	0	0	0	0
Residue spreading	Loading (pumping)	0	0	0	0	0
	Transport to fields	0	0	0	0	0
	Storage in satellite storage	0	0	0	0	0
	Spreading	0	0	0	0	0
Production steps	Cultivation	567	567	652	652	700
	Harvest	54	54	48	48	126
	Storage	118	118	120	120	125
	Transport	45	45	41	41	52
	Total	**785**	**785**	**861**	**861**	**1004**
Production means	Machinery	355	355	425	425	376
	Diesel	78	78	106	106	93
	Electricity	37	37	40	40	47
	Heat	50	50	54	54	63
	Liming	5	5	5	5	5
	Seeding material	70	70	70	70	43
	Pesticides	33	33	33	33	149
	Plastic	0	0	0	0	0
	Buildings	31	31	26	26	16
	Fertilizer	125	125	102	102	212
	Total	**785**	**785**	**861**	**861**	**1004**

Table 52. Economic costs [€/ha].

		Sugarbeet	
	Scenario	**C1:c**	**C1:m**
	Year	**2**	**2**
Material	Fertilizer N	115	115
	Fertilizer P	50	50
	Fertilizer K	100	100
	Seeds	234	234
	Pesticides	253	253
	Liming	5	5
Operations	Stubble treatment (cultivator)	25	25
	Ploughing	108	108
	Harrowing	27	27
	Sowing	56	56
	Rolling	24	24
	Fertilizer spreading	17	17
	Spraying	77	77
	Inter-row cultivation	45	45
	Harvest of beets	216	216
	Field transport	83	83
	Loading to beet cleaner	13	13
	Beet cleaner	16	16
	Transport of dirt	0	0
Storage	Plastic cover for stack	0	0
Residue spreading	Loading (pumping)	0	0
	Transport to fields	0	0
	Storage in satellite storage	0	0
	Spreading	0	0
Production steps	Cultivation	1165	1165
	Harvest	216	216
	Storage	99	99
	Transport	118	118
	Total	**1598**	**1598**
Production means	Machinery	604	604
	Diesel	139	139
	Electricity	0	0
	Heat	0	0
	Liming	5	5
	Seeding material	234	234
	Pesticides	253	253
	Plastic	99	99
	Buildings	0	0
	Fertilizer	265	265
	Total	**1598**	**1598**

Appendix H. Methods for the assessment of environmental impact and socioeconomic value

H.1 LIFE CYCLE ASSESSMENT

The assessment of environmental impact was based on the ISO standard 14044 for life cycle assessment (LCA) (ISO, 2006). The life cycle inventory (LCI) outcomes were characterized into three impact categories where ecological consequences where considered relevant for the study; global warning potential (GWP), eutrophication potential (EP) and acidification potential (AP). The equivalency factors (category indicators) are shown in Table 53.

Table 53. Equivalence factors applied in the present study

g per g	**GWP100**[a] CO_2-eq	**AP**[b] SO_2-eq	**EP**[c] PO_4^{3-}-eq	**EP**[c] PO_4^{3-}-eq
Emission to	*air*	*air*	*air*	*water*
Carbon dioxide, CO_2	1			
Methane, CH_4	25[b]			
Nitrous oxide, N_2O	298[c]		0.27	
Nitrogen oxides other than N_2O, NO_x [d]		0.7	0.13	
Sulphur dioxide, SO_2		1		
Ammonia, NH_3		1.88	0.35	0.35
Ammonium NH_4^+				0.33
Nitrate (NO_3^-) and nitrite (NO_2^-)			0.1	0.1
Nitrogen (N)			0.42	0.42
Phosphate (PO_4^{3-})			1	1
Phosphorus (P)			3.06	
COD[e]				0.022

[a] (IPCC, 2007), In the calculations according to EU RED (EU, 2009), equivalence factors of 23 (CH4) and 296 (N2O) are specified, originating from (IPCC, 2006). In the sensitivity analysis, updated GWP20 factors (IPCC, 2013) where evaluated, 86 (CH_4) and 268 (N_2O).
[b] (Wenzel and Hauschild, 1998) in (CML, 2015)
[c] (Guinée et al., 1992) in (CML, 2015)
[d] For NOx, the characterization factor is calculated assuming the chemical formula NO_2
[e] COD is an indicator used to measure organic material which gives the total amount of organic matter that can be oxidized by chemical oxidation. Emissions where sometimes given as BOD, which refers to the biodegradable share of COD, and that was converted to EP using an equivalence factor 0.044, based the assumption that BOD represents 50% of COD.

IPCC presented GWP characterization factors for methane (CH_4) and nitrous oxide (N_2O) in the guidelines of 2006 (IPCC, 2006). These factors have since been updated (IPCC, 2007, 2013), but the factors from 2006 on a 100 year horizon are still applied in the methodology for greenhouse gas impact for biofuels according to the renewable energy directive (EU, 2009). The 2007 factors are used e.g. in the Swedish national inventory report (SEPA, 2015) and are used in the base case in calculations in the present study. The factors published in 2013 are evaluated in a sensitivity analysis, where also the impact from a shorter time perspective (20 years) is chosen, which increases the GWP of methane (IPCC, 2013), Table 53.

AP and EP equivalency factors are based on stoichiometric relations. For AP, the potential for generation of hydrogen ions (H^+) per kg substance in relation to H^+ generation per kg SO_2 is calculated, reflecting the maximum contribution to acidification that a compound can cause (Baumann and Tillman, 2004) and is here given as SO_2-equivalents (SO_2-eq). For EP, the impact on potential build-up of aquatic biomass is calculated. The emitted nutrient is then assumed to be limiting for growth, while other compounds are in unlimited supply, and the biomass generated is assumed to have the formula $C_{106}H_{263}O_{110}N_{16}P$ (Guinée *et al.*, 1992). The EP is here given as PO_4^{3-} equivalents (PO_4^{3-}eq.).

For particles, the different sources used for emission data had different level of detail on particle size distribution. Often, particle emission was just given as particulate matter (PM) without further details. If a size distribution was given, only particles of a size up to 2.5 μm (PM2.5) were included, since it is these small particles that have the greatest impact on health.

The assessment in the cereal regions was performed with focus on the use of arable land and with a functional unit of 1 ha of arable land as average for the crop rotations studied in the different scenarios. For the greenhouse gas emissions, the results were also recalculated per MJ of biofuel, which was defined as compressed vehicle gas at filling station.

The inventory data was based on identifying relevant input data for current practice and level of technology. In the LCA, the fuel cycle emissions (including end use emissions) for energy use and input materials were included. Production and maintenance of vehicles, roads, buildings and farm machinery were excluded. Special efforts were made in calculation of the mass flows of nitrogen species and soil organic carbon (SOC). SOC was modelled in great detail, as were the nitrogen flows in crop residues, added mineral fertilizer and biofertilizer. However, the general emissions factor applied in converting nitrogen to an emission of N_2O is a rough estimate (IPCC, 2006; SEPA, 2015). As described in (SEPA, 2015), activities related to agricultural soil management is a major source of N_2O emissions and -when applying a general emission factor- will also be the emission where uncertainties are large. In the Swedish inventory report of 2015, 4 out of 10 emissions that are described as having great uncertainties are related to N_2O from arable land (N_2O from inorganic N-fertilizer, N_2O from crop residues, N_2O from manure fertilization and N_2O emissions or avoided emissions due to mineralization or uptake during loss or gain in SOC). Choice of emission factors was based on the IPCC 2006 Guidelines (IPCC, 2006), and IPCC default emission factors were used if not national emission factors were available in the Swedish national inventory report (SEPA, 2015).

For the calculation of nitrogen leakage from arable land, introduction of changes in crop rotations, with changes in fertilizer application including biofertilizer was evaluated regionally and on crop rotation level applying a calculation model from the Swedish board of agriculture (SJV, 2015a).

H.2 SYSTEM BOUNDARIES

The assessment included cultivation, harvest and storage of crops, transport of grass as biogas feedstock, biogas production, upgrading, compression and distribution, as well as storage, transport and application of the digestate as biofertilizer on arable land. For the cereal regions, emissions in the scenarios defined as "current" were calculated and used as reference emissions for calculation of the impact of a change to the scenarios under investigation, described as "modified". A systems expansion approach, in accordance with the recommendation in the ISO standard of LCA (ISO, 2006), was applied. In the systems expansion, the total output of grains and oil seed was equivalent in the different scenarios. Thus, a reduced output of grains and oil seeds on a crop rotation level, due to the introduction of grass cultivation in the modified scenarios, was compensated for by additional grain and oil seed production outside the farm. This additional cultivation was assumed to take place nationally, and no impact of indirect land use changes due to displacement effects was included. The biogas produced in the modified scenarios was assumed to replace diesel in heavy vehicles.

H.3 RENEWABLE ENERGY DIRECTIVE

The calculations for the cereal regions per MJ biofuel were also performed with the calculation methodology defined based on the EU renewable energy directive (EU RED) (EU, 2009). The method in the directive was interpreted as in Swedish law (HBL, 2010) together with guidelines from the Swedish Energy Agency (SEA, 2011, 2012). This methodology does not include a full LCA, and excludes e.g. the impact of SOC of a change in crop rotations. Also, the method of assessing the impact of a change, where a reference scenario is used, is not applied. Emissions from the production of machinery, buildings, infrastructure etc. are not to be included.

For the livestock region, only calculations based on EU RED was performed. To allocate costs and environmental impact between manure and grass, typical values for different feedstock types were used in a mass balanced based calculation (Hagberg, 2011).

H.4 QUANTIFYING THE SOCIOECONOMIC VALUE

The environmental impact of the investigated scenarios was also quantified as socioeconomic value. Literature data on socioeconomic evaluations was used, as summarized in Table 14.

The carbon dioxide tax is a current value from Swedish decision makers, and is also the value recommended for use for socioeconomic evaluation of CO_2-emissions by Trafikverket (2016). This group behind the annual update of socioeconomic values related to transport express that there is no realistic alternative which is better than a value based on this politically agreed tax value, which for 2016 is 0.12 € (kg CO_2)$^{-1}$ (Skatteverket, 2016). The higher value suggested for sensitivity analyses is 0.37 € (kg CO_2)$^{-1}$ (Trafikverket, 2016). For acidification potential, the low value given is only the local impact, while for the high also regional impact is added (Trafikverket, 2016). For particle emissions, the given value is for PM2.5, particles with a size of <2.5 μm. The value differs much between countryside (62 € kg^{-1}) and small or larger cities, where examples given for cities with 36 000 – 120 000 inhabitants range from 341-626 € kg^{-1}, and emissions in Stockholm city are given a value of 1 390 € kg^{-1}. Here, an average value for the smaller cities is chosen, 484 € kg^{-1} for emissions in cities. For eutrophication, a suggested range of values from The Swedish Environmental Protection Agency are used (SEPA, 2009), and recalculated to the monetary value of 2016 (SCB, 2016c).

Appendix I: General data for emissions and environmental impact

In this section, the selected background data for the calculation of emissions and environmental impact are summarized, together with a discussion about the selection.

I.1 EMISSIONS RELATED TO ENERGY USE

The energy use in domestic road transport in Sweden per fuel in 2014 is shown in Figure 46. Fuel use in domestic road transport, Sweden 2014. Data shown as energy (LHV) in PJ a-1.. The figure shows that the main fuel used was diesel, which together with blend in of the renewable fuels fatty acid methyl esters (FAME) and hydrogenated vegetable oils (HVO) together represents 59%. Petrol with blend in of bioethanol represents 38%. Pure FAME, ethanol and biogas (with or without blend with natural gas) are still used in minor amounts.

A low-blend of ethanol and FAME limited to 5% (based on volume) has, in the EU fuel quality directive been allowed since 2006. In 2014 the volume FAME in diesel was 4.9% and the ethanol in petrol was 4.8% (Figure 46). However, the relatively new fuel HVO, which has been used in Sweden since 2011, and where the production is rapidly increasing, is not limited by similar restrictions. No limits for low-blend levels exist in the fuel quality directive, and the tax exemption for HVO is not linked to a maximum low-blend limit, so the share of HVO in diesel has steadily increased from <1% in 2011 to 8.3% in 2014 (SEA, 2015c). No emission data for the use of this rapidly appearing diesel mix is, however, yet available, and data for the use of diesel with a 5% blend of FAME is used for production and distribution both for diesel use in heavy vehicles and tractors, and in the systems expansion, where biogas replaces fossil fuels (Table 54). For biogas, only end use emissions are included (Table 54).

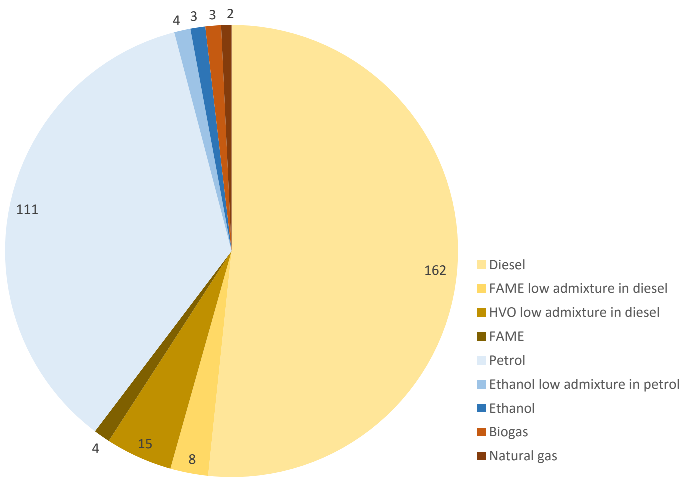


Figure 46. Fuel use in domestic road transport, Sweden 2014. Data shown as energy (LHV) in PJ a^{-1}.

Table 54. Emissions at road transport and in tractor operations. Representative values for present vehicle fleet including both production, distribution and end use. Emissions to air if not else stated.

[mg MJ^{-1}] if not else stated	CO_2 [g MJ^{-1}]	NO_X	SO_2	Parti-cles	CH_4	N_2O	NH_3	NH_4^+	NO_3	PO_4^{3-}
Diesel with 5% RME, heavy vehicle/bus[a]	76.8	673	17	14	34	1.8	0.6			
emission to water							26	<0.1	<0.1	<0.1
Petrol with 5% ethanol, light vehicle[a]	76.0	170	15	2.2	49	2.5	29	29	29	29
emission to water							<0.1	<0.1	-2.9	<0.1
Diesel with 5% RME, tractor [a,b]	75.9	818	19	12	34	2.0	0.7	0.7	0.7	0.7
emission to water							26	<0.1	<0.1	<0.1
Biogas, heavy vehicle[c]	0	200	0	0.5	114	0	na			
Biogas, light vehicle[c]	0	10	0	0.5	39	0	na			

na: not available
[a] Fuel cycle emission as average for heavy vehicles, city and long distance busses (Gode *et al.*, 2011)
[b] End use emissions for tractor operations are average for harrowing, sowing, tilling, fertilization, harvest, loading and field transports (Börjesson *et al.*, 2010).
c Emissions only at end use (Börjesson *et al.*, 2010). End use methane emissions are from Göthe (2013), and represent typical emissions from existing biogas vehicle fleet.

In the systems expansion, when biogas is assumed to replace diesel in heavy vehicles/busses, a correction for end use efficiency is done. 1.18 MJ biogas is assumed to be needed to replace 1 MJ diesel.

Greenhouse gas (GHG) emissions are also calculated according to the methodology in the renewable energy directive. There, the GHG emission should be calculated as g CO_2-eq MJ-1 (LHV) and the reduction given against a given GHG emission for fossil fuels of 83.8 g CO_2-eq MJ^{-1} (EU, 2009). The emissions at end use of the renewable fuel should be set to zero. When this methodology is used, the specified emission of 83.8 g CO_2-eq MJ^{-1} for fossil fuels is also used for transports and tractor operations in the production system, replacing the values in Table 54.

Other emissions related to energy use are summarized in Table 55. In the calculations according to the renewable energy directive, the GHG regional emission for electricity is specified as Nordic average electricity, with an emission of 34.9 g CO_2-eq MJ^{-1} (SEA, 2011; Martinsson *et al.*, 2012), which is replacing the emission for electricity shown in Table 55. For the biogas plant in the livestock scenario, the specific conditions at the biogas plant used as model was used. Here district heating is used for biogas upgrading, and the value for Sävsjö district heating grid from 2014 of 13 g CO_2-eq MJ^{-1} heat was used in the calculations (SF, 2016).

Table 55. Emissions to air at energy use.

Emissions [g MJ^{-1}]	CO_2	NO_x	SO_2	Particles	CH_4	N_2O	NH_3
Nordic average electricity [a]	19	0.040	0.038	0.013	0.067	0.002	0.004
Wood chips [b]	3.3	0.10	0.04	0.003	0.005		
Oil (EO1) production & distribution [c]	5.3	0.019	0.013	0.001	0.029	<0.001	<0.001
Oil (EO1) end use large scale [c]	74.3	0.20	0.025		0.001	0.001	
Oil (EO1) end use small scale [d]	77.4	0.102	0.050	0.005			

[a] Average NORDEL 2013-2015. Average for production mix and only including high voltage transmission losses (Ecoinvent, 2016).
[b] Fuel cycle emissions per MJ fuel (Börjesson *et al.*, 2010). For heat from wood chips in the biogas plant a heat efficiency of 85% is assumed.
[c] Emissions of ammonium, nitrate and phosphate to water <0.0001 g MJ-1 and are excluded. Per MJ fuel (Gode *et al.*, 2011).
[d] Per MJ fuel (Börjesson and Gustavsson, 1996). For heat from oil in cereal drying a heat efficiency of 85% is assumed.

I.2 EMISSIONS FROM THE PRODUCTION OF MINERAL FERTILIZER.

The emissions from the production of mineral fertilizer are based on data given by (Börjesson *et al.*, 2010), Table 56. Most of the mineral N sold in Sweden is in the form of calcium ammonium nitrate (52%) (SJV, 2015c). A large share of the GWP from the production of ammonium nitrate will be due to the emissions of N_2O in production, which was presented as 15 g (kg N)$^{-1}$ without catalytic cleaning and 3 g (kg N)$^{-1}$ with catalytic cleaning (Börjesson *et al.*, 2010). More recent data presented by Yara shows similar values, N_2O emission with best available technology (BAT) within EU (catalytic abatement) are 4.2 g (kg N)$^{-1}$, while average emission is 17 g (kg N)$^{-1}$ (Fossum, 2014). The data originally presented by Börjesson (2010) are based on the assumption that 30% of the production applies catalytic removal of N_2O. The GWP for N would then be 6.6 kg CO_2-eq (kg N)$^{-1}$. This was later updated with an assumption about improvements where catalytic cleaning was assumed to occur for 50% of the production, giving a GWP of 6.0 kg CO_2-eq (kg N)$^{-1}$ (Börjesson and Tufvesson, 2011). In a more recent study, it has been pointed out that many installations in Western Europe have lately implemented catalytic cleaning of N_2O, which is said to represent 60% of the total GWP for the average western European mineral fertilizer production (Ahlgren *et al.*, 2015). The GWP presented by Yara (BAT) is 3.6 kg CO_2-eq (kg N)$^{-1}$, while ammonium nitrate produced based on Russian energy efficiency and without catalytic cleaning for N_2O is given as 8.1 kg CO_2-eq (kg N)$^{-1}$ (Fossum, 2014).

Kool *et al.* (2012) have presented data showing higher greenhouse gas emissions than reported in Table 56, much due to updates related to higher methane leakage from the natural gas grids. Natural gas is used both as fuel (1/3) and as the fossil feedstock used to obtain the hydrogen (2/3) needed for ammonia synthesis (IFIA, 2009). The GWP from the production of ammonium nitrate in western Europe is presented as being 8.0 kg CO_2-eq (kg N)$^{-1}$, with a global average of 9.5 kg CO_2-eq (kg N)$^{-1}$.

The data presented by Börjesson and Tufvesson (2011) gives a GWP that is in between the BAT-values given by the leading Swedish supplier of mineral fertilizer (Fossum, 2014) and the typical production in western Europe (Kool *et al.*, 2012; Ahlgren *et al.*, 2015) and also provides information on emissions other than GHG, and are chosen for the present study. The data given for P (Table 56) corresponds to a GWP of 3.2 kg CO_2-eq (kg P)$^{-1}$, which is in the same range as the more recently updated value of 3.4 kg CO_2-eq (kg P)$^{-1}$ (Kool *et al.*, 2012). For K, the GWP given by Kool et al is 1.6 kg CO_2-eq (kg K)$^{-1}$, so higher that the corresponding emission based on the data in Table 56, but since that study does not present information on emissions other than greenhouse gasses, the data presented by Börjesson *et al.* (2010) is used also for the production of K (Table 56).

Table 56. Emissions at the production of mineral fertilizers

[g kg^{-1}]	CO_2	NO_X	SO_2	Particles	CH_4	N_2O
N [a, b]	3200	8	4.6	0.82	3.1	9
P [a]	2900	18	39	9.5	7.2	0.29
K [b]	440	2,7	5.9	1.4	1.1	0.002

[a] (Börjesson *et al.*, 2010)
[b] (Börjesson and Tufvesson, 2011)

For calculations according to the Renewable Energy Directive, emission data based on Ahlgren *et al.* (2011) are applied, who have calculated typical values for Swedish crops for biofuels. The emissions for mineral N, however, are instead chosen from Fossum (2014), who give BAT values for mineral N delivered to Sweden, Table 57.

Table 57. GWP for the production of mineral fertilizers used in the EU RED calculations

[kg CO_2-eq kg^{-1}]	
N [a]	3.1
P [b]	0.71
K [b]	0.46

[a] (Fossum, 2014)
[b] (Ahlgren *et al.*, 2011)

I.3 OTHER INPUTS IN CULTIVATION

Other relevant inputs in cultivation are seed, liming, pesticides, ensiling additive and the plastic used for ensiling.

For seed production, the share of seed used in cultivation in relation to the actual grain harvest (or assumptions about a typical seed production) is calculated. The actual calculated emission in the LCA for each specific crop is then used together with this ratio to calculate the seeding emission (Strid and Flysjö, 2007).

Limestone contains 12% of C (weight), which is calculated to be lost as CO_2 as a worst case scenario according to the methodology in (IPCC, 2006). This gives an emission of 0.44 kg CO_2-eq (kg limestone)$^{-1}$. The emissions of the application are included under the total diesel consumption in crop production.

For pesticides, a general emission per active substance was used based on Ahlgren *et al.* (2011). For ensiling additive, the compound Promyr was assumed to be used, and aggregated data given by the producer was used (GWP: 0.72 kg CO_2-eq kg^{-1}, EP: 0.68 mg PO_4^{3-}-eq kg^{-1}, AP_ 7.05 mg SO_2-eq kg^{-1} (Flysjö *et al.*, 2008). Plastic was assumed to have an energy content of 86 MJ kg^{-1}, and to be burnt after use in large scale incineration using the same emissions as for oil (EO1), Table 55.

I.4 CROP PRODUCTION IN SYSTEMS EXPANSION

In the systems expansion, the crops lost from the crop rotations at introduction of grass in the cereal regions should be included by adding the emissions of production elsewhere. Here, general and typical emission data was sought after, and not the case specific data (involving biofertilization, SOC change etc) calculated in the present study. For GWP, EP and AP, data for winter wheat and oil seed rape was found, and the emission for wheat was used for all cereals (Table 58). The GWP data are comparable to the data given by Ahlgren *et al.* (2011), where also data on oats are presented. No similar general data was found for particle emissions on crop DM basis, and here own data from cultivation in the current crop rotations with mineral fertilizer was used. Particle emissions are not influenced by field related emissions, but are related to input of material and energy, so is assumed to be more generally applicable.

Table 58. Emission data for crop production used in the systems expansion

	GWP [a] [kg CO_2-eq (t DM)$^{-1}$]	EP [a] [g PO_4^{3-}-eq (t DM)$^{-1}$]	AP [a] [g SO_2-eq (t DM)$^{-1}$]	Particles [b] [g (t DM)$^{-1}$]
Winter wheat	245	105	565	73
Oil seed rape	485	1 831	1 357	130
Spring barley	*	*	*	77
Oats	*	*	*	77

[a] Based on recalculation of data given by Börjesson *et al.* (2010).
[b] Based on own data recalculated per crop DM and given as average for both regions

I.5 NITROUS OXIDE EMISSIONS IN RELATION TO SOIL MANAGEMENT

For field emissions, the emission factor for N_2O was 1% emission of N_2O-N of added N for all types of N-addition (crop residues, mineral fertilizer, biofertilizer). The indirect emission of N_2O from N leakage to water was 0.75%, and of N released to to air 1%, all according to the estimated emission factors presented by (IPCC, 2006).

In addition, soil organic matter is assumed to contain N in a ration of 1:10 to the C content. Thus a degradation of soil organic matter will give a release of N in soil, which is assumed to contribute to N_2O formation in the same way as the other N-additions in soil above (SEPA, 2015). In the same way, N is assumed to be incorporated in organic matter at a ratio of 1:10 to C in SOC build-up, making this share of N unavailable for N_2O formation.

I.6 AMMONIA EMISSIONS FROM FERTILIZER APPLICATION

Ammonia emissions at field application are assumed to be 0.9% of added N for mineral fertilizer (SEPA, 2015). The application of biofertilizer with trailing shoes has been shown to put the fertilizer below the crop, and in under some conditions also give a soil incorporation impact (SJV, 2005). For spreading of liquid manure in cereals early summer with trailing hoses (without shoes), a typical emission has been presented as 7% of added TAN (Karlsson and Rodhe, 2002). Danish studies has shown that with shallow incorporation, the emission can be reduced to 50% of that of trailing hose for pig manure before and after biogas production (Hansen, 2010), and data for application in whinter wheat in spring with trailing shoes have given emssions of less than 1% (Karlsson and Rodhe, 2002). The selected emission of NH_3-N for early summer application in winter wheat is 3% of added TAN.

In biofertilizer application in grass, the emissions of ammonia are high in conventional trailing hose application, 50% of TAN can be typical for early summer according to (Karlsson and Rodhe, 2002). Danish studies for liquid manure present lower values, 14-19% with trailing hose application in spring and 18-25% in summer (Hansen, 2001), which are reduced to 10-14% and 13-18% respectively if the manure is incorporated in the soil at application (Miljöministeriet, 2010). Another Danish study has shown that application in a shallow furrow will reduce the emission compared to trailing hose without soil incorporation with 20-80%, and values of between 10-32% emission of NH_4-N of TAN are presented (Hansen, 2001). The base case value for emission of NH_3-N for early summer application in grass is difficult to choose based on the given data, but is selected as 20% of added TAN. However, it has been shown that new technologies for biofertilizer injection both in grass and cereals can reduce ammonia emissions to nearly zero (Hansen, 2010). In the sensitivity analysis, a decreased emission to 10% of added TAN in grass is evaluated to illustrate the importance for AP and EP impacts of minimizing NH_3 losses. This type of technology could, however, also give an increased risk for N_2O emissions (Hansen, 2010).

I.7 NITROGEN LEAKAGE FROM ARABLE LAND

The calculations of nitrogen leakage made with the model Vera (SJV, 2015a) were made based on the cultivation conditions presented in Appendix B. The calculations were made on municipality-basis, and in the cereal regions, 8 municipalities distributed over the selected regions were selected. Average leakage for the crop rotations were 41.9 (C1:c), 39.9 (C1:m), 30.1 (C2:c) and 29.4 (C2:m) kg N ha^{-1} a^{-1}. The results for the individual crops are shown in Table 59. The mean value for the region for each scenario is used in the LCA. In the calculations according to RED, the individual values for grass in Scenario C1:m and C2:m are used.

For the livestock region, only leakage data for one municipality (Sävsjö, where the biogas plant is located) was evaluated (Table 60), and the individual data for the different years for grass were used.

Table 59. Leakage of NO_3^- given as kg NO_3-N ha^{-1} a^{-1} for selected municipalities

		Lund	Landskr.	Tomelilla	Kristianst.	Skurup	Ängelh.	Trelleb.	Ystad	**Mean**
C1:c	Winter wheat	46	45	39	39	49	42	41	45	43
	Sugar beets	37	36	31	31	39	34	33	36	35
	Spring barley	40	40	33	34	43	36	36	40	38
	Winter wheat	46	45	39	39	49	42	41	45	43
	Spring barley	47	46	40	40	50	43	42	46	45
	Winter oilseed rape	51	50	44	43	55	47	46	50	48
C1:m	Winter wheat	48	47	41	41	51	44	43	47	45
	Sugar beets	37	36	31	31	39	34	33	36	35
	Spring barley	40	40	33	34	43	36	36	40	38
	Winter wheat	48	47	41	41	51	44	43	47	45
	Grass full prod. year	23	22	18	19	24	20	20	22	21
	Grass break year	59	57	52	50	62	55	52	57	55
		Lidköp.	Trollh.	Lilla ed.	Skara	Skövde	Tidah.	Falköp.	Herrlj.	**Mean**
C2:c	Winter wheat	28	34	44	27	29	27	30	28	31
	Spring oat	26	32	42	25	27	26	28	27	29
	Winter wheat	24	29	40	23	25	24	26	25	27
	Winter wheat	28	34	44	27	29	27	30	28	31
	Spring barley	30	36	47	29	31	29	32	30	33
	Spring oat	26	32	42	25	27	26	28	27	29
C2:m	Winter wheat	30	36	46	29	31	28	32	29	33
	Spring oat	26	32	42	25	27	26	28	27	29
	Winter wheat	25	31	41	25	26	25	27	26	28
	Winter wheat	29	36	46	28	30	28	31	29	32
	Grass full prod. year	16	19	26	15	16	16	17	16	18
	Grass break year	34	41	51	32	35	31	36	32	36

Table 60. Leakage of NO_3^- given as kg NO_3-N ha^{-1} a^{-1} for Sävsjö municipality and scenario L:m

Crop	
Spring oat	48
Grass yr 2	25
Grass yr 3	25
Grass yr 4, break year	62

GRASS FOR BIOGAS – ARABLE LAND AS A CARBON SINK

Det finns regioner i Sverige där vi i dag tappar organiskt material i åkermark på grund av ökad specialisering, intensifiering och minskad användning av biogödsel. I spannmålsdominerade områden där djurtätheten är låg visar beräkningar att det kol som tappas bidrar till koldioxidutsläpp som är nästan fyra gånger så stora som utsläppen av växthusgas från dieselanvändningen i odling i samma område. För att vända denna ohållbara utveckling krävs en ökad tillförsel av kol, vilket kan ske genom att tillföra odlingsrester eller genom biogödsling.

Den här rapporten visar att genom att införa vallodling av gräs skulle kolförlusten från åkermark stoppas eller till och med kunna vändas och bidra till minskade växthusgasutsläpp, både i odling och i transportsektorn. Det kan därför vara samhällsekonomiskt motiverat att uppmuntra denna förändring även om det är negativt sett ur andra miljöaspekter. Vallodling kan också vara ett sätt att på lång sikt öka eller bibehålla en hållbar livsmedelsproduktion lokalt och regionalt. Det måste förstås vägas mot det faktum att åkermark används för odling av energigrödor.

Dagens EU-regler för beräkning av klimatnyttan för producerad biogas tar inte hänsyn till markkolsförändringar, och inte heller till den positiva effekten av att ersätta mineralgödsel med rötrest.

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