REPORT f3 2013:27

IMPACT OF BIOGAS ENERGY CROPS ON GREENHOUSE GAS EMISSIONS, SOIL ORGANIC MATTER AND FOOD CROP PRODUCTION

– A CASE STUDY ON FARM LEVEL

Report from an f3 project

Authors:

Lovisa Björnsson, Mikael Lantz and Pål Börjesson
Environmental and Energy Systems Studies, Lund University

Thomas Prade and Sven-Erik Svensson
Biosystems and Technology, Swedish University of Agricultural Sciences

Håkan Eriksson
E.ON Gas Sverige AB

THE SWEDISH KNOWLEDGE CENTRE
FOR RENEWABLE TRANSPORTATION FUELS

PREFACE

This report is the result of a cooperation project within the Swedish Knowledge Centre for Renewable Transportation Fuels (f3). The f3 Centre is a nationwide centre, which through cooperation and a systems approach contributes to the development of sustainable fossil-free fuels for transportation. The centre is financed by the Swedish Energy Agency, the Region Västra Götaland and the f3 Partners, including universities, research institutes, and industry (see www.f3centre.se).

The authors acknowledge valuable contribution from farm inspector Hans Rasmussen, Wrams Gunnarstorps farm, on crop yields on the case farm. Furthermore, we would like to thank Thomas Kätterer and Martin Bolinder, Department of Ecology, SLU Uppsala, for support with the soil carbon model, the data used for its calibration and for valuable comments on the results.

This report should be cited as:

Björnsson, L. *et al*., (2013) *Impact of biogas crop production on greenhouse gas emissions, soil organic matter and food crop production–A case study on farm level*. Report No 2013:27, f3 The Swedish Knowledge Centre for Renewable Transportation Fuels, Sweden. Available at www.f3centre.se.

EXECUTIVE SUMMARY

Soil degradation is a widespread problem: erosion, loss of soil organic matter and compaction are some of the degradation processes that are threatening soil fertility throughout the EU (Soilservice, 2012). Intensively cultivated clay soils have in Swedish studies been shown to give up to 20% decreasing food crop harvest yields due to soil compaction and reduced soil organic matter content (Arvidsson & Håkansson, 1991).

The remedy is a combination of improved farm machine technology and increased soil organic matter content. This can be achieved by;

- replacing mineral fertilizers by manure or other biofertilizers containing organic matter, such as digestate (the effluent from biogas production), and
- changing the crop rotation to include e.g. ley, green manuring crops and catch crops.

In assessing the climate benefits of energy crops on arable land it is thus important to also consider the effects on the cultivation system and long term soil fertility. In an analysis of climate effects of changed agricultural practice and crop rotations, increased soil organic matter content will have a dual effect. The build-up of soil organic matter has been shown to be positively correlated to most soil ecosystem services (Soilservice, 2012). In addition, the long term carbon sequestration will reduce greenhouse gas (GHG) emissions (Anderson-Teixeira et al., 2009; Röing et al., 2005).

In this project, a case where the biogas process potentially could contribute to more efficient land use by maximizing positive crop rotation effects and by supplying a biofertilizer on farm land, was investigated in a farm based case study. The purpose was to evaluate a scenario where a biogas plant has been integrated in an agricultural region with mainly stockless farming and intensively cultivated clay rich soils. Ley crops are introduced in crop production and used as biogas feedstock. The biogas plant also provides biofertilizer. The project also includes a systems analysis of the biogas plant as presently operated. The farm based case study also includes an analysis of possible scenarios for the farm scale system, and the impact on farm level of the introduction of biogas production and crops on arable land which act as biogas feedstock. The overall objective of the project was to analyse how the integrated production of food crops and energy crops for biogas production impacts the GHG emissions per land area, the soil organic matter and the total crop output. On the farm that is the model for this study, soil compaction on the medium to heavy clay soils is a problem. The crop yields are 5–20% lower than average yields for the region. Aware of the problem, three years of meadow fescue for seed production has been integrated in the cereal based crop rotation on the most problematic soils. However, the market for grass seeds being limited, the economic possibilities for integrating ley crops in other parts of the crop rotation is limited in a region with little demand for cattle feed. The approach in the farm based case study was to integrate 1-2 years of ley crops in the crop rotation and to use this as feedstock for biogas production. The effects of this on GHG emissions, soil organic matter and food crop production was evaluated.

The project contains of two parts. In the first part a life cycle assessment (LCA) was performed for the biogas plant at Söderåsens Bioenergi, that is presently in operation and that is located within the boundaries of the farm investigated in the farm based case study. Since the full results of this study have been scientifically published elsewhere (Lantz & Börjesson, 2013), the present report is a

mere summary and the information relevant for the second part, the farm based case study, is summarized.

In the second part of the project, the farm based case study was performed. The farm delivers already today one biogas feedstock to the biogas plant, manure from pigs. Thus, the case study was split in two parts; in the first part the introduced change is that pig manure is used for biogas production instead of being used as biofertilizer directly. This reflects a change that has already occurred. In the second part, ley is introduced in the present crop rotation, as described above, and used in addition to manure as biogas feedstock, a situation that can potentially occur in the future. This two stage approach allows separate assessment of the effects of introducing these two different biogas feedstocks on the farm based system. Data regarding agricultural aspects of the analysis were inventoried, e.g. crop rotations, harvest yields, soil properties, energy in- and output and emissions in biomass production. The functional unit was set to 1 hectare (ha) of arable land. The assessment included cultivation, harvest and storage of crops, manure storage, biogas production, upgrading and compressing, digestate storage and application and soil carbon changes. Data from the LCA of the biogas plant was used for the biogas production part of the farm based case study. The assessment applied a systems expansion approach, in accordance with the recommendation in the ISO standard of LCA (ISO, 2006). In the systems expansion, the total output of grains (wheat and oats) and oil seed (rape seed) is equivalent in the different scenarios. Thus, a reduced output of grains and oil seeds on a farm level, due to the introduction of ley crop cultivation, was compensated for by additional grain and oil seed production outside the farm. This additional cultivation was assumed to take place within the region on excess farmland, not leading to any indirect land use changes due to displacement effects. The output of upgraded biogas delivered to the natural gas grid, was assumed to replace fossil vehicle fuel.

The reference scenarios in the farm based case study include the conventional handling of manure that took place before the biogas plant was established (Scenario A) or where biogas was produced only from manure (Scenario B). Scenario A is the reference system for Scenario B. For Scenarios C1-3, where ley crops are produced and used for biogas production, Scenario B is used as the reference scenario.

The climate benefit was shown to be high for all the investigated scenarios where ley is introduced in the crop rotation and used for biogas production. Introducing a change where 20-33% of the 650 ha farm is used for production of ley as a biogas crop, will give avoided GHG emissions of 1 240-1 500 kg CO_2-eq per ha, yr. before systems expansion. In a systems expansion, the production of the biofuel (biogas) and the lost crop production is included in the analysis. Compensating the lost crop production decreases the climate benefit of the systems. Even so, the resulting reduction in GHG emission is large. The resulting net avoided GHG emissions after systems expansion are 2.2 to 3.2 t CO_2-eq per ha, yr. on average for this 650 ha farm. The emission reductions are also calculated as avoided emission per GJ fuel produced and utilized for replacing fossil vehicle fuels to enable comparison with GHG emissions for other vehicle fuels and amount to -88 to -107 kg CO_2-eq per GJ fuel used. The reference GHG emission for fossil fuels in the renewable energy directive is 84 kg CO_2-eq/GJ. The effect of introducing ley for biogas production at a farm and using it as biogas energy crop will thus give a biofuel with an emission reduction of 106-128% when replacing fossil fuels. This can be compared to the emission reduction of 90% presented in part one of this project (Lantz & Börjesson, 2013) when biogas produced from a mix of manure and industrial residues replaces fossil fuels, or the reduction of 159% when only manure is used for biogas production in scenario B in the farm based case study. Emission reductions above 100% indicate that the production itself, not only the utilization of the biofuel to replace fossil fuels, gives avoided emissions. When using only manure for biogas production, the avoided emissions of methane and N_2O during storage and after soil application when the manure is handled as digestate are the main causes for the avoided GHG emissions from biogas production. When introducing ley in a cereal based crop rotation, the main cause of the avoided emissions from production is the soil carbon build up, both from the crop residues from ley in the crop rotation, and from the carbon-rich digestate that is recirculated at the farm.

The climate benefit for scenarios with ley production is to a large extent the effect replacing fossil fuels with the biogas produced, 480-870 l petrol/ha, yr. or 15-24 TJ/yr. over the whole 650 ha farm. Equally important is the effect on increased soil organic matter content on farm level. Apart from the role as a carbon sink and the impact on GHG emissions, the increase in soil carbon levels is important for long term soil fertility and productivity on this type of compacted clay soils. In the ley scenarios, the soil carbon content increases steadily from 2% today to 3% within 20–30 years to reach a steady state level of 4–5 %. Here, the possibility of using ley for biogas production opens up for a possibility of integrating ley in the crop rotation in cereal intensive areas even if there is no demand for cattle feed. Still, land would be taken out of food production. The impact of increased soil organic matter on soil fertility and the potential of increasing yields could partly compensate for this. A yield increase of 10% would partly counteract the loss of grain and oil seed in the scenarios with ley in the crop rotations. In this study, however, the loss in food crop production is from a climate perspective compensated for by adding GHG emissions for crop from the additional grain and oil seed cultivation outside the farm to fulfil an unchanged total output of food products. One important aspect in all ley scenarios is that the ley is undersown the year before the main harvest (called year 0), making ley biomass harvest possible in autumn year 0. This gives good land use efficiency, using the benefit of harvesting that extra biomass. The economic feasibility of this small harvest remains to be evaluated. The sensitivity analysis shows that the ley based systems are sensitive to the chosen data for calculation of amounts of crop residues. However, the IPCC data set, which gives very high straw amounts for cereals, is not valid for the actual conditions in the Nordic countries, and the calculation method developed within the project is considered to give a better estimation of actual conditions. The ley scenarios are also sensitive to calculations assuming high methane leakage from the digestate storage. The emissions evaluated in the sensitivity assessment are however high considering that the digestate is used as fertilizer in the period 1 April-15 May, and the share of the annual digestate production that is stored during the warm part of the year, May-October, is less than 1/3 of the annual production. An aspect like this, the time for spreading, is important to consider when a digestate containing high amounts of organic matter, like in this case from ley crops with relatively low biodegradability, is produced.

The soil carbon contribution is important to consider in all systems where biomass is removed from farm land. Part of the evaluation was also to assess the effect of using manure for biogas production, and then recycling the residue, the digestate, to the farm as biofertilizer. The climate benefit for this manure based biogas production is very good, but it has been argued that the impact on soil organic matter could be negative since much of the easy degradable carbon is removed as biogas when digesting manure. However, the negative effect on soil carbon was in this study shown to be small; the impact on soil carbon change in the long perspective is negligible. The main positive impact of biogas production from manure is when the biofuels replaces fossil transportation fuels, but the impact on reduced biogenic emissions of N_2O and methane is also important.

In the study it has also been shown which features of the investigated scenarios that are most important for good GHG efficiency. At the same time, effects on soil organic matter content and food crop yields have been presented. The outcome is important in fulfilling the future criteria in sustainability certification of biofuel systems, such as avoiding indirect land use changes and maximizing GHG performance (Ahlgren & Börjesson, 2011). Both outcomes are equally important for improved understanding of scenarios involving soil fertility challenges and land use competition between food and energy crop production.

Neither biogas production from manure nor from ley crops shows good profitability from a biogas plant perspective at present biofuels prices (Lantz et al., 2013), which might hinder the introduction of such systems in spite of the positive impact on GHG emissions. The intention of the researchers behind the present study is to follow up with a study encompassing several Swedish regions and including economic evaluations of ley as biogas feedstock, including aspects that are important from the farm perspective as effects of preceding crop and soil carbon contribution.

SAMMANFATTNING

Intensivodlade lerjordar har i Sverige visats kunna ge 20% lägre skördar p.g.a. av markpackning och låga mullhalter (Arvidsson & Håkansson, 1991). Denna typ av problem med försämrad bördighet har även uppmärksammats på andra håll inom EU (Soilservice, 2012). En del av lösningen är att höja mullhalten i marken, vilket kan uppnås bl.a. genom förändrade växtföljder, där t.ex. gräs/klövervall integreras i växtföljden. En annan lösning är att ersätta mineralgödsel med organiska gödselmedel. I områden där djurhållning inte är så vanligt förekommande är dock tillgången på stallgödsel låg, likaså avsättningen för vall som djurfoder.

I detta projekt studeras effekten av förändringar i odlingssystemet vid en gård där markpackning är ett problem på de intensivodlade lerjordarna. Skördenivåerna för höstvete, havre och raps vid gården har uppskattats ligga 5–20% lägre än normalskördarna i regionen. Avsättningen för vall som djurfoder i regionen är låg, liksom tillgången på stallgödsel som organiskt gödselmedel. I denna studie undersöks därför effekterna av att introducera 1–2 år gräs/klövervall i växtföljden och använda vallen för biogasproduktion istället för som djurfoder. Restprodukten efter biogasproduktion, rötresten, återförs till åkermarken som organiskt gödselmedel. Effekten på växthusgasemissioner för odlingssystemet, markkolshalten samt produktionen av livsmedelsgrödor utvärderas på gårdsnivå. Det övergripande syftet är att visa på effekterna av denna typ av integrerad produktion av livsmedels- och energigrödor, för att nyansera bilden av att energigrödor på åkermark ofta presenteras som en motsättning till livsmedelsproduktion. Möjligen har ingen hänsyn tagits till vikten av bibehållen eller ökad mullhalt (markkolshalt) och den komplicerade och dubbla effekt ökad mullhalt har vid en analys av klimatpåverkan från förändrade odlingssystem. Ökad mullhalt har visat sig vara korrelerad till de flesta markekosystemtjänsterna och markens långsiktiga bördighet (Soilservice, 2012) och dessutom bidrar långsiktig kolinbindning till en direkt klimatnytta (Anderson-Teixeira et al., 2009; Röing et al., 2005).

Projektet består av två delar, där den första utgör en systemstudie av biogasanläggningen vid Söderåsens Bioenergi. Denna biogasanläggning är i drift sedan 2007, och ligger inom Wrams Gunnarstorps marker, den gård som utvärderats i den gårdsbaserade fallstudien. Denna del av arbetet har publicerats vetenskapligt (Lantz & Börjesson, 2013), så i denna rapport återges endast en sammanfattning samt de delar som är relevanta för del två, fallstudien på gårdsnivå.

Studiens andra del är den gårdsbaserade fallstudien. Inom gården finns redan idag ett organiskt gödselmedel, svingödsel, som används för biogasproduktion innan restprodukten, rötresten, sprids på åkermark. För att tydligt separera effekten av svingödsel och effekten av att introducera vall har utvärderingen gjorts i två steg. I steg ett jämförs direkt användning av svingödsel som organiskt gödselmedel i odling (Scenario A) med förfaringssättet att först använda gödseln för biogasproduktion och därefter rötresten som gödselmedel i odling (Scenario B). I steg två introduceras vall i livsmedelsväxtföljden på olika sätt för att sedan används för biogasproduktion. Rötresten återförs till åkermarken (Scenario C1-C3). I analysen av växthusgasemissioner används systemutvidgning, där den minskade produktionen av livsmedelsgrödor då vall introduceras ersätts med odling av dessa grödor som antas ske inom regionen på tillgänglig åkermark, d.v.s. inga indirekta markanvändningsförändringar antas ske. Den biogas som produceras antas uppgraderas och levereras på naturgasnätet, och därefter ersätta fossila drivmedel.

I steg ett, där den konventionella livsmedelsbaserade växtföljden odlas med tillförsel av orötad (Scenario A) eller rötad (Scenario B) svingödsel, är produktionen av grödor densamma. I det studerade odlingssystemet, där all tillförsel av organiska gödselmedel är viktig för mullhaltsuppbyggnad, är det dock centralt att analysera effekten på markkolstillförseln om stallgödseln rötas innan den används som biogödsel. Det beror på att en del av kolet i stallgödseln avgår i form av biogas då gödseln rötas. Effekten på den långsiktiga markkolsuppbyggnaden visades här dock vara mycket liten, där markkolshalten stabiliseras på 3,0 resp. 2,9% efter 145–150 år. Emissionen av växthusgaser från odlingssystemet påverkas positivt. Enbart emissionen från odling är 2,4 t CO_2-ekv/ha, år, och sjunker till 2,1 när biogasproduktion inkluderas. I systemutvidgningen, när den producerade biogasen antas ersätta fossila drivmedel, sjunker emissionen ytterligare, till 1,7 t CO_2-ekv/ha, år. Klimatnyttan av att övergå från odling med gödsling med svingödsel till systemet med biogasproduktion samt återförsel av rötad gödsel till åkermark ger alltså en undviken emission på 0,7 t CO_2-ekv/ha, år med bibehållen livsmedelsproduktion och försumbar påverkan på markkolshalten.

I steg två där 1–2 år vall integreras i växtföljden är emissionerna från systemen (vilket även inkluderar biogasproduktion av gödsel/vall) 0,6 till 0,9 t CO_2-ekv/ha, år, vilket ska jämföras med 2,1 t CO_2-ekv/ha, år från den konventionella växtföljden med enbart biogasproduktion från gödsel. Här är det främst den ökade markkolsinbindningen som ger emissionsminskningen. När sedan i systemutvidgningen bortfallet av grödor ersätts, och producerad biogas ersättare fossila drivmedel, är växthusgasemissionerna från vallscenarierna -0,6 till -1,5 t CO_2-ekv/ha, år. Klimatnyttan av att gå från den livsmedelsbaserade 4-åriga växtföljden till en ny växtföljd där vall inkluderas på 20–33% av åkermarken och där denna vall används för biogas, blir alltså -2,2 till -3,2 t CO_2-ekv/ha, år. Denna klimatnytta är till stor del en effekt av att den producerade biogasen, motsvarar 480–870 l bensin per ha, ersätter fossila drivmedel. Lika viktig är dock den ökade långsiktiga markkolsinbindingen. Förutom funktionen som kolsänka är markkolsinbindningen viktig för långsiktig markbördighet, och en viktig åtgärd för att komma åt bördighetsproblem på kompakterade lerjordar. I vallscenarierna ökar markolshalten från 2% idag till 3% efter 30 år, för att nå 4–5% efter 150 år. Möjligheten att få avsättning för vall för biogasproduktion kan skapa en drivkraft för att integrera vall i växtföljden i områden där efterfrågan på vall som foder är liten. Åkermark tags då ur livsmedels/foderproduktion, men effekten av ökad markkolshalt på bördighet kompenserar för en del av detta bortfall.

Resultaten från detta projekt är viktiga för att visa hur hållbarhetskriterier för produktion av biodrivmedel från åkermark, såsom att undvika indirekt förändrad markanvändning och att maximera växthusgasreduktioner, kan uppfyllas (Ahlgren & Börjesson, 2011). Det är viktigt för förståelsen av system som omfattar bördighetsutmaningar och konkurrerande markanvändning mellan livsmedels- och energigrödor på åkermark.

Produktion av biogas som drivmedel från gödsel och vall visar dålig lönsamhet med dagens drivmedelspriser, vilket kan leda till att dessa råvaror inte används för biogasproduktion i så stor utsträckning trots den goda klimatnyttan (Lantz et al., 2013). Forskarna bakom detta projekt avser att följa upp med en analys som omfattar ekonomiska utvärderingar av vall för biogasproduktion, med fokus på aspekter som förfruktsvärde och markkolsuppbyggnad som är viktiga ur odlarperspektiv.

CONTENTS

1 INTRODUCTION

What can be the impacts when biogas energy crops are introduced at a farm? The evaluation is a complex task due to the major effects a change in crop rotation has on the cultivation system. Thus it is difficult to generalize, introduction of different crops on different land types will result in different impacts. To meet future sustainability criteria for biofuel systems based on energy crops on farmland, it will be crucial to demonstrate supply systems with minor effect on current food production. One potentially interesting strategy could be to improve the soil productivity, and thereby food crop yields, through dedicated and integrated food and energy crop rotations. The hypothesis behind this study was that the introduction of ley crops as bioenergy crops on intensively cultivated land could be a case where positive synergies could be achieved due to the positive impact on soil organic matter content and reduced soil compaction. In a case where the market possibilities are small for ley as fodder, the ley would be used for biogas production, generating a biofertilizer that could be returned to arable land. The use of arable land for other purposes than for food/feed production can then be balanced against the benefits of reduced GHG emissions and soil improvement.

Soil degradation is a widespread problem: erosion, loss of soil organic matter and compaction are some of the degradation processes that are threatening soil fertility throughout the EU (Soilservice, 2012). Intensively cultivated clay soils have in Swedish studies been shown to give up to 20% decreasing food crop harvest yields due to soil compaction and reduced soil organic matter content (Arvidsson & Håkansson, 1991). Within the recently finalized EU project Soilservice it has been concluded that current arable farming practices in the EU imply that soil biodiversity will continue to decline and crop yields will be lower than if biodiversity was well maintained. Soil management that builds up soil carbon - which is a good proxy for soil biodiversity - will both improve the sustainability of food production and farmers' incomes (Soilservice, 2012).

The remedy is a combination of improved farm machine technology and increased soil organic matter content. This can be achieved by;

- replacing mineral fertilizers by manure or other biofertilizers containing organic matter, such as digestate (the effluent from biogas production)
- changing the crop rotation to include e.g. ley, green manuring crops and catch crops

In assessing the climate benefits of energy crops on arable land it is thus important to also consider the effects on the cultivation system and long term soil fertility. In an analysis of climate effects of changed agricultural practice and crop rotations, increased soil organic matter content will have a dual effect. The build-up of soil organic matter has been shown to be positively correlated to most soil ecosystem services. In addition, the long term carbon sequestration which will reduce greenhouse gas emissions (Anderson-Teixeira et al., 2009; Röing et al., 2005). In the present analysis, a case was outlined where the biogas process potentially contributes to more efficient land use by maximizing positive crop rotation effects and by supplying a biofertilizer on farm land where this has a strong positive impact on increasing soil carbon content and potentially the fertility. The effects on GHG emissions, soil organic matter and food crop yields when introducing biogas production from manure and ley crops will be compared and discussed.

In the study it is clarified which features of the investigated scenarios that are most important for good GHG efficiency. At the same time, effects on soil organic matter content and food crop yields are presented. The outcome is important in fulfilling the future criteria in sustainability certification of biofuel systems, such as avoiding indirect land use changes and maximizing GHG performance (Ahlgren & Börjesson, 2011). Both outcomes are equally important for improved understanding of scenarios involving soil fertility challenges and land use competition between food and energy crop production. This knowledge is important for bioenergy companies in their future commercial strategies regarding biomass supply systems that fulfil coming sustainability criteria, and for authorities and policy makers in their development and implementation of adequate regulations and incentives.

1.1 BACKGROUND

The background to the proposed farm based case study is the increased awareness of the relevance of soil compaction and carbon stock in relation to biofuel production on farm land that has been described by e.g. Börjesson et al. (2010) and Lantz et al. (2009).

The project builds on, and could not have been performed without, previous research activities at Lund University (LU) and The Swedish University of Agricultural Sciences (SLU). EON Gas Sverige AB (EON) as the industrial partner has actively supported biogas research, and has interest in improved knowledge about manure and crop based biogas systems. The outcomes of the project will also be useful for other stakeholders with similar interests. Data from two research studies previously funded by EON, a systems analysis of a biogas plant funded by The Swedish Energy Agency and long term field trials/fertility studies performed within SLU provide the background data that has made it possible to conduct this assessment of a crop rotation integrating food and biogas crops. EON has financed a 35 ha energy crop cultivation trial (2009-2011) on land where soil compaction was an issue. The land is a farm with mainly clay rich soils in North West Skåne (Wrams Gunnarstorp) which is close to Ekebo, the location for a long-term fertility experiment performed by SLU. This trial will provide background data for soil carbon modelling. The energy crop cultivation trials have been performed by SLU, and the purpose was to provide data on energy crop yields. The cultivation trials were performed with fertilization with digestate from a biogas plant within the farm, Söderåsens Bioenergi. A systems analysis of the biogas plant at Söderåsens Bioenergi was performed by LU in 2009, funded by The Swedish Energy Agency (Lantz et al., 2009). In addition, EON financed a project on improved biogas production in waste-based biogas production by energy crop addition (2008-2011), which was performed by LU (Nges et al., 2011). EON also owns additional data, where LU has experimentally determined methane yields from the energy crops from the SLU cultivation trial described above, which will be used in the present analysis. In addition, EON will provide updated operational data from the Söderåsens Bioenergi biogas plant to be used in the present study.

1.2 PURPOSE AND OBJECTIVE

The purpose of this project was to evaluate a scenario where a biogas plant has been integrated in an agricultural region with mainly stockless farming and intensively cultivated clay rich soils. Ley crops are introduced in crop production and used as biogas feedstock. The biogas plant also provides biofertilizer. The project includes a systems analysis of the biogas plant as presently operated. It also includes a farm scale case study, assessing the impact on farm level of the introduction of biogas production and crops on arable land which act as biogas feedstock.

The overall objective is to analyse how the integrated production of food crops and energy crops for biogas production impacts the GHG emissions per land area, the soil organic matter and the total crop output.

1.3 METHOD

The project contains two parts. In the first part a life cycle assessment (LCA) was performed for the biogas plant Söderåsens Bioenergi, presently in operation and located within the boundaries of the farm to be investigated. The analysis was based on the previously mentioned study, Lantz et al. (2009), which was updated with data from full scale operation in 2011. The assessment included transport of feedstock, biogas production, upgrading, compression and distribution via the natural gas grid, as well as storage, transport and application of the digestate as biofertilizer on arable land. The assessment applied a systems expansion approach, in accordance with the recommendation in the ISO standard of LCA (ISO, 2006). In the reference system, where no biogas is produced, petrol is assumed to be used as vehicle fuel. The reference system also includes the conventional handling of manure and sludge that took place before the biogas plant was established, as well as the utilization of some vegetable residues as animal feed. The digestate produced in the biogas plant is assumed to replace mineral fertilizers. The system expansion regarding the replacement of mineral fertilizers by digestate includes the avoided production of mineral fertilizers, the increased input of organic matter into the soil and the increased risk of soil compaction due to heavier field equipment. Changes in the carbon levels in the soil and soil compaction will affect the soil fertility and thereby crop yields, leading to indirect effects on energy and GHG performance in crop cultivation on the farm in question. Calculations of GHG emissions and energy balance were based on the ISO standard 14044 for LCA (ISO, 2006). The functional unit (FU) was set to 1 MJ upgraded and compressed vehicle gas distributed via the natural gas grid. The full results of this study have been scientifically published elsewhere (Lantz & Börjesson, 2013), so in the present report only a summary, and the information relevant for the second part, the farm based case study, is presented.

In the second part of the study, the farm-based case study is performed. This case study is in turn performed in two steps. The farm already delivers one biogas feedstock to the biogas plant, namely manure from pigs. In the first step, the studied change is that pig manure is used for biogas production instead of being used as biofertilizer directly. This reflects a change that has already occurred at the farm. In the second step, ley is introduced in the present crop rotation and is used in addition to manure as biogas feedstock, a situation that can potentially occur in the future. This two-step approach allows separate assessment of the effects of introducing these two different biogas feedstocks within the farm based case study. Data regarding agricultural aspects of the analysis were inventoried, e.g. crop rotations, harvest yields, soil properties, energy in- and output and emissions in biomass production. An LCA was performed with the functional unit 1 hectare (ha) of arable land. The assessment included cultivation, harvest and storage of crops, manure storage, biogas production, upgrading and compressing, digestate storage and application and soil carbon changes. Life cycle inventory data from the LCA performed for the biogas plant in part one of the project was used as input for the biogas part of the farm based case study. The assessment applied a systems expansion approach, in accordance with the recommendation in the ISO standard of LCA (ISO, 2006). In the systems expansion, the total output of grains (wheat and oats) and oil seed (rape seed) is equivalent in the different scenarios. Thus, a reduced output of grains and oil seeds on a farm level, due to the introduction of ley crop cultivation, was compensated for by additional grain and oil seed production outside the farm. This additional cultivation was assumed to take place within the region on excess farmland, not leading to any indirect land use changes due to displacement effects. The output of upgraded biogas delivered to the natural gas grid, was assumed to replace fossil vehicle fuel, or petrol.

The reference systems include the conventional handling of manure that took place before the biogas plant was established (Scenario A) or where biogas was produced only from manure (Scenario B). Scenario A is the reference system for Scenario B in step one of the farm based case study. For Scenarios C1-3, where ley crops are produced and used for biogas production in step two, Scenario B is used as the reference scenario.

2 THE FARM

2.1 BACKGROUND

The farm based case study investigates the effects of different crop rotation and biomass utilization scenarios. As a basis for this case study, a farm with mainly clay rich soils in North West Skåne was chosen. This farm, Wrams Gunnarstorp, is located close to the site of long-term soil carbon field trials in Ekebo, performed by SLU (Kirchmann et al., 1999).

The farm operates on an area of approx. 700 hectares (ha), of which about 50 ha are light sandy soils. This study concentrated on the 650 ha of medium to heavy clay soils with soil clay content up to 65%.

The soils are rather cold, and crop establishment is often carried out very shortly in the autumn after harvest of the previous crop Table 2.1. Establishment is rather slow, and the risk for the plants to be too big for overwintering is little. However, this leaves no opening for the introduction of after-sown catch crops, but under-sown catch crops can be a possibility in winter wheat followed by spring oats next year (see more information about crop rotation in section 2.2). Traditionally these soils were plowed at a depth of approx. 15-20 cm followed by 4-5 harrowing steps. Since 2005, plowing is replaced with subsoiling/cultivation to a depth of 25-30 cm in combination with a light compaction of the top soil layer (false seedbed), followed by 2 steps of harrowing. The subsoiling lifts the soil and opens it up, but does not blend down the humus material. Oxygen and water can penetrate the soil reducing the risk of pools of standing water, else resulting in no crop growth. Crop establishment was improved and has resulted in yield increases. In this study, it was assumed that the soil is ploughed conventionally as was done on the farm before 2005 in order to exclude the effects of this change of soil treatment.

2.2 CROP ROTATION

On the major part of the farm, a 4-year crop rotation typical for the region is used, Table 2.1.

Table 2.1. Common crop rotation and typical sowing and harvest dates on the Wrams Gunnarstorp farm.

Year	Crop	Sowing date	Harvest date
1	Winter rapeseed	1-10 August	20 July
2	Winter wheat	1-20 September	10 August
3	Winter wheat	1-20 September	10 August
4	Oats	1-20 April	20 August

Due to the problems associated with heavy clay soils and in order to improve soil fertility, a second crop rotation is operated on a smaller fraction of the farm, including three years of meadow fescue for seed production, Table 2.2. The area on which this improved crop rotation is used is limited by the marketing possibilities for meadow fescue seeds.

Table 2.2. Improved crop rotation and typical sowing and harvest dates on the Wrams Gunnarstorp farm.

Year	Crop	Sowing date	Harvest date
1	Winter rapeseed	1-10 August	20 July
2	Winter wheat	1-20 September	10 August
3	Winter wheat	1-20 September	10 August
4	Oats	1-20 April	20 August
5	Meadow fescue	as undersown crop in oats	10 July
6	Meadow fescue		10 July
7	Meadow fescue		10 July

The 4-year crop rotation as shown in Table 2.1 is assumed to be used on the 650 ha that this study was based on.

2.3 STRAW BOILER

In 2000, an 800 kW straw boiler was constructed at the farm in order to produce heat for cereal drying and floor heating of all farm and estate buildings. About 1000 bales of approx. 410 kg dry weight are collected annually. On the farm, the ash is returned to the field at a dosage of approx. 2 t per hectare for nutrient recycling, but only about 4-5 t per year are produced. Only wheat straw and straw from meadow fescue are used.

In the farm based case study, the straw utilization was limited to represent the amount of energy needed for cereal drying only. This amount of straw was subtracted from the scenarios and the remaining straw was assumed to be chopped and incorporated into the soil. The amount of ash in the different scenarios was less than 5 t and therefore handling and effect of the ash was excluded.

2.4 LIVESTOCK AND MANURE PRODUCTION

The farm has a history of livestock production. Milk production was phased out in 1960. From 1995 onward, pigs were produced in a stable with 1600 fattening places, producing approx. 3500 t of manure annually. In 2009, another pig stable with 900 places was acquired, adding another 2000 t of manure. The manure that was formerly used as a biofertilizer on the heavy clay soils of the farm is today delivered to the biogas plant, and the digestate (the liquid residue from the biogas plant) has replaced the untreated manure as biofertilizer. The initiative to building the biogas plant within the farm boundaries was partly based on the need for more biofertilizer than the existing pig manure.

2.5 SOIL CARBON CONTENT

The latest analysis of soil carbon content expressed as humus on the Wrams Gunnarstorp farm dated from 1984. That year, the soils had an average humus content of 3.8% and consisted of very heavy clay soils, heavy clay soils and medium clay soils. The fairly high content of humus in the Wrams Gunnarstorp soils at this time was probably a result of the consequent use of cow manure as biofertilizer until 1960. In this study a starting value of 4% of humus content (2% of soil carbon) content was assumed.

2.6 BIOGAS PLANT

In 2006, the Söderåsen Bioenergi biogas plant was taken into operation. The plant operates on the pig manure produced on the farm and slaughterhouse waste from the facility that slaughters the farm's pigs. These feedstocks, however, only add up to a minor part of the total feedstock addition. The biogas plant is described in more detail in Chapter 4.

The digestate produced at this biogas plant is used as biofertilizer at the farm since 2007 and has replaced a large share of mineral fertilizer that was used on the farm prior to the construction of the biogas plant. The amount of digestate produced in the biogas plant (close to 50 000 t/yr. in 2011) is enough for fertilization of about 1 400 ha and is now used as biofertilizer both at the Wrams Gunnarstorp farm and neighboring farms. The benefits of this biofertilizer utilization are presented in the biogas plant assessment (the first part of the project, Lantz & Börjesson, 2013). In the farm based assessment, however, the amount of biofertilizer in the form of digestate is calculated to correspond only to the product from digestion of the feedstocks supplied by the farm, the pig manure and the ley. Thus, the benefits of adding digestate originating from other organic feedstock (food industrial waste etc.) is here excluded.

3 THE SCENARIOS

In part two of this project, the farm based case study, strategies to improve soil organic matter content and reduce GHG emissions on the case farm were investigated in two steps. The purpose was to evaluate if a biogas plant can provide organic fertilizer needed for soil productivity improvement. Also, ley crops were evaluated in different crop rotations in terms of its potential to improve soil carbon content and therefore fertility. These ley crops were used as biogas feedstock. The stepwise evaluation was enabled by defining scenarios where one change at a time was introduced.

3.1 BASIC ASSUMTIONS

In order to simplify this systems analysis, certain assumptions and limitations have been set that are valid for all scenarios tested. The farm is assumed to be limited to the 650 ha of heavy clay soil. Pig production itself is excluded, but delivers pig manure from the two stables into the system and the system boundary includes the manure storage. Straw is removed from the fields for use as solid fuel in the straw boiler. The amount of straw removed represents the amount needed to dry the cereals produced and therefore varies between scenarios. Fixed crop-specific fertilization levels were used throughout all scenarios. Where required, manure or digestate application was complemented with mineral fertilizer applications. The composition of the digestate in scenarios B and C was calculated based on the farm-related feedstocks delivered to the biogas plant (manure and ley crops), so is not the actual digestate from the biogas plant (with addition of food industrial waste etc.). The reason for this is that the analysis of the scenarios should reflect only the system effects of the changes at the farm. The environmental impact of the production and use of the digestate as it occurs at the biogas plant (part one of the project) is described in the paper by Lantz & Börjesson (2013).

3.2 SCENARIO A

In this scenario, the presently used conventional 4-year crop rotation (Table 2.1) with (1) winter rapeseed, (2) winter wheat, (3) winter wheat and (4) oats has been maintained. 6 050 t/yr. of liquid pig manure is stored in an open, naturally crust covered, liquid manure storage tank and subsequently used as organic fertilizer, reflecting the manure handling before introduction of the biogas plant in 2006 (Figure 3.1).

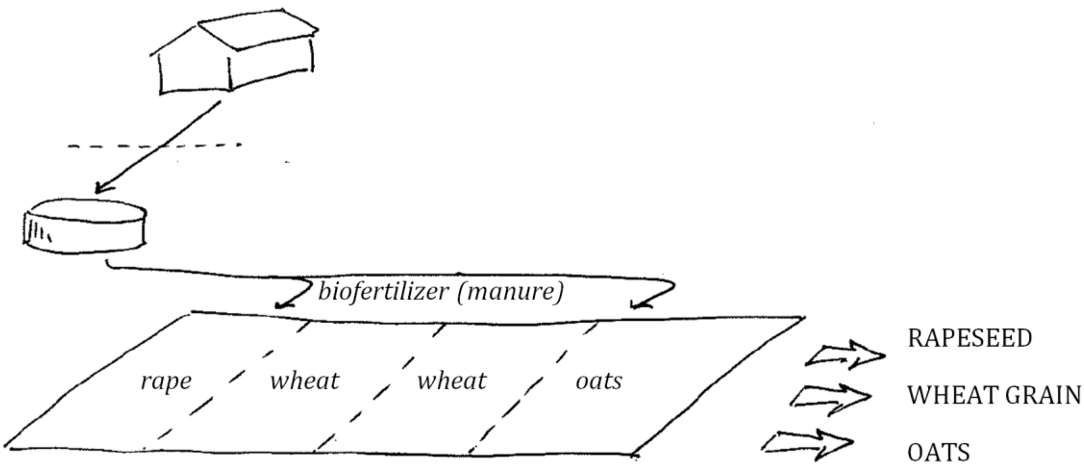


Figure 3.1. Crop rotation and food crop production Scenario A. Pig manure is stored under crust cover in open tank and used as biofertilizer.

3.3 SCENARIO B

In scenario B, the crop rotation remains the same as in scenario A, but the pig manure is pumped to and treated in the biogas plant (Figure 3.2). The effluent from manure digestion, the digestate, is stored in a covered digestate storage tank and subsequently recycled within the farm as biofertilizer. The biogas produced from the manure is upgraded, compressed, spiked with propane and delivered to the natural gas grid.

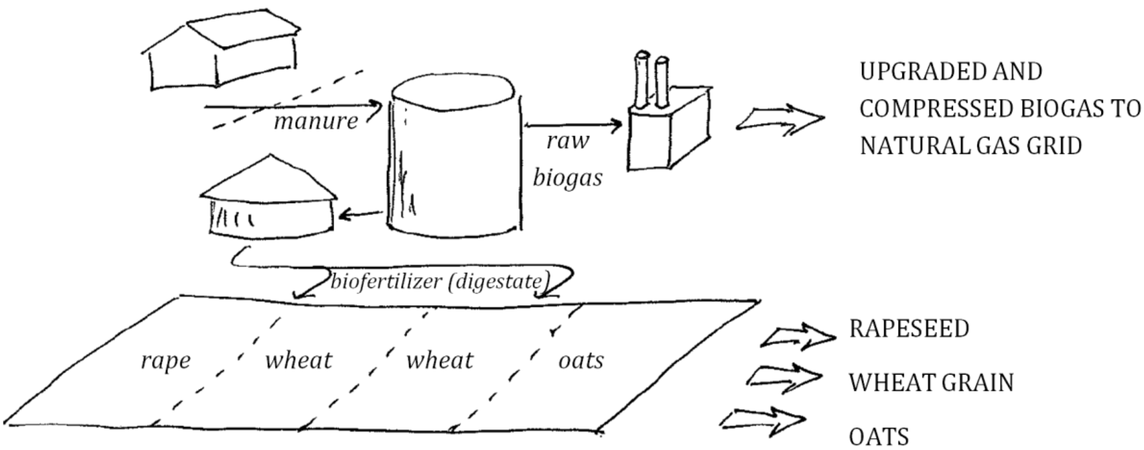


Figure 3.2. Crop rotation and food crop production Scenario B. Pig manure is added to the biogas plant and biogas is produced. Digestate from manure digestion is used as biofertilizer.

3.4 SCENARIO C

In the three C scenarios, the crop rotation is changed to also include ley crops for biogas production. Ley is introduced in three different ways, resulting in scenarios C1, C2 and C3. The ley is harvested, pre-treated, added to the biogas plant (together with the pig manure described in scenario B), and the digestate is stored in a covered digestate storage tank and subsequently recycled to the farm as biofertilizer. The biogas produced from the manure and the ley is upgraded, compressed, spiked with propane and delivered to the natural gas grid.

Scenario C1

In scenario C1, one year of ley crops is implemented in the crop rotation following oats (Figure 3.3).

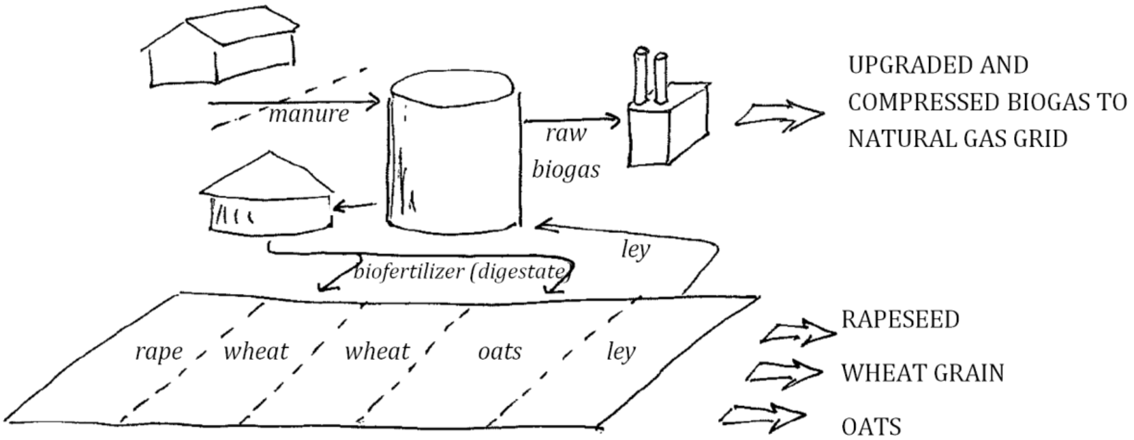


Figure 3.3. Crop rotation and food crop production Scenario C1. Pig manure and ley from one year in the five year crop rotation is added to the biogas plant and biogas is produced. Digestate from manure and ley digestion is used as biofertilizer.

Scenario C2

In scenario C2, two consecutive years of ley crops are implemented in the crop rotation following oats (Figure 3.4).

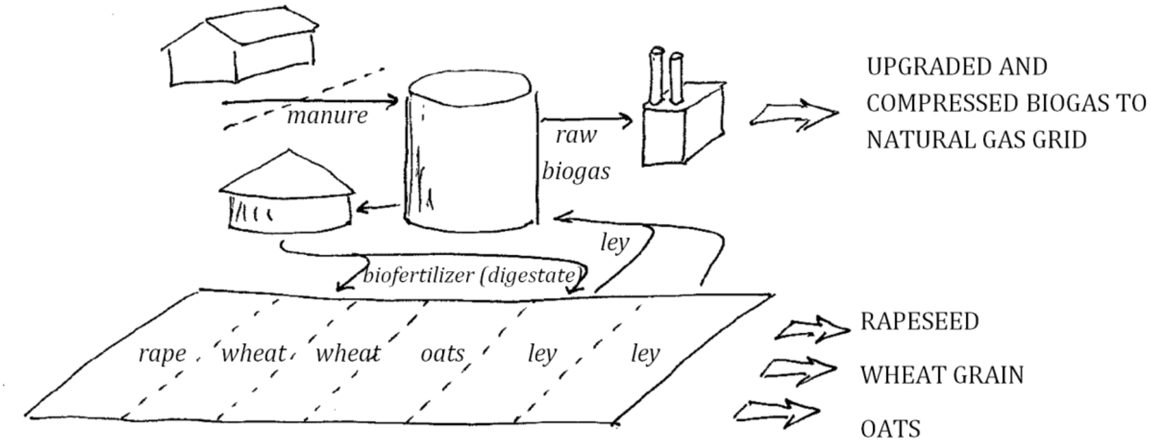


Figure 3.4. Crop rotation and food crop production Scenario C2. Pig manure and ley from two years in the six year crop rotation is added to the biogas plant and biogas is produced. Digestate from manure and ley digestion is used as biofertilizer.

Scenario C3

In scenario C3, one year of ley crops are implemented in the crop rotation by replacing oats (Figure 3.5).

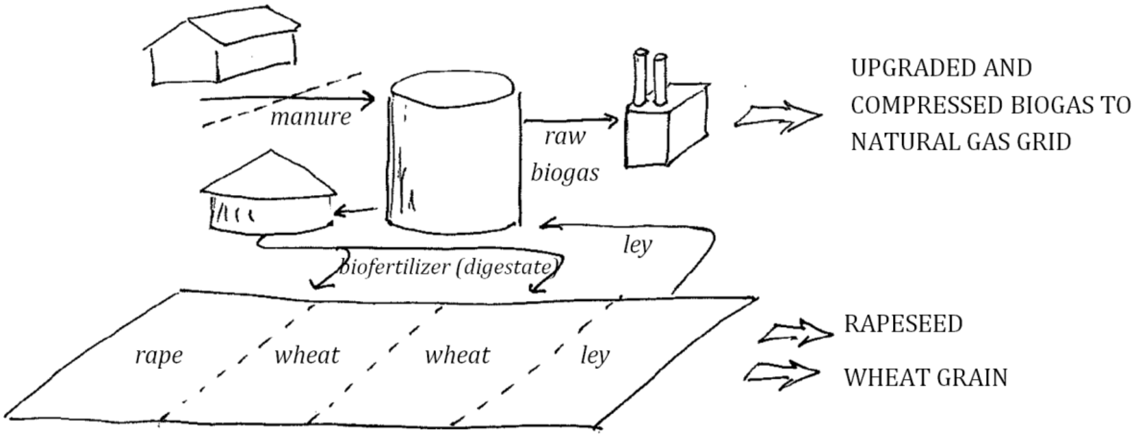


Figure 3.5. Crop rotation and food crop production Scenario C3. Pig manure and ley from one year in the four year crop rotation is added to the biogas plant and biogas is produced. Digestate from manure and ley digestion is used as biofertilizer.

4 BIOGAS PRODUCTION

In the present project, LCA's from two different perspectives were included; the biogas plant (part one) and the farm based (part two) perspective. In part one, the assessment of the biogas plant, the co-digestion plant at Söderåsens Bioenergi as operated in 2011 was evaluated. Here, the pig manure produced at the investigated farm is co-digested with manure from other sources and with various food industry waste. The biogas, 90 TJ/yr., is upgraded and utilized as a vehicle fuel, distributed via the natural gas grid. The supply of feedstock to this co-digestion plant in 2011 was 50 900 t. The energy use and the emissions of GHG were investigated from a systems perspective. The main outcomes from the biogas plant assessment (part one of the project) are repeated in section 4.1. The full result is presented separately as a scientific paper by Lantz & Börjesson (2013). The data from part one of the project that are used in the second part, the farm based case study, are presented in section 0.

4.1 SUMMARY OF RESULTS FROM THE BIOGAS PLANT PERSPECTIVE

Biogas produced from food industry waste and manure in a modern co-digestion plant could, according to the results presented by Lantz & Börjesson (2013) reduce GHG emissions by approximately 90% when the biogas produced replaces fossil vehicle fuel. Crucial factors for the GHG performance is the use of natural gas for generation of process heat and emissions of methane, especially from the upgrading process. The utilization of digestate also results in GHG benefits when mineral fertilizers are replaced. There is, however, a risk of some drawbacks from this replacement in the form of increased soil compaction due to heavier field equipment. This risk could be minimized by applying appropriate spreading technology. The replacement of mineral fertilizers by digestate will also lead to increased input of long-term stable organic matter to the soil, leading to increased soil carbon content. In total, the estimated decrease in yields represents due to soil compaction is 2.1 g CO_2-eq./MJ biogas or 25% of net GHG emissions from the biogas system. The addition of soil carbon reduce GHG emissions with 0.7 g CO_2-eq./MJ.

In the analysis, several possible technical improvements to further reduce GHG emissions were identified as well. However, the economic prerequisites of the specific improvements varied, from profitable from a business perspective to unprofitable from a socio-economic point-of-view. The most favourable improvement was to produce process heat using wood chips instead of natural gas which was found to be profitable from a business perspective.

4.2 BIOGAS PRODUCTION IN THE FARM LEVEL SCENARIOS

To calculate the role of the biogas production from the farm perspective, data from the assessment of the co-digestion plant was used. In the analysis of the farm, the pig manure and ley were assumed to be added to that existing biogas plant for co-digestion with other feedstocks. This is in reality the case for the pig manure today. The GHG emissions for biogas production and upgrading based on the conditions in 2011 are used, and energy input and emissions are in the farm scale assessment allocated to the different feedstocks. Some effects are based on the added weight of feedstock, some on the volume of biogas or upgraded methane that is produced from that specific feedstock. The relevant background data are described below, and summarized in Table 4.2

Feedstock and biogas amounts

In scenario B, the pig manure is used for biogas production. For pig manure, the amount of manure produced in scenario A and stored in an open (no roof cover) concrete tank (6050 t/yr.) includes rain water corresponding to 10% of the manure amount. When the manure is pumped directly from the stable to the biogas plant, rain water dilution is avoided, and the amount is 5500 t/yr. In scenarios C1-C3, one or two years of ley were included in the crop rotation at 650 ha. The ley yields in the different scenarios are presented in Appendix B, Table B1. The ley is field dried to 35% dry matter (DM) at harvest, and no losses other than water are assumed to occur. The ley is then transported to the biogas plant where it is stored in bunker silos, and a loss of 5% of DM during ensiling is subtracted. The individual composition of ley from different harvests is given in Table A3 in Appendix A. Methane yields for the feedstocks are discussed in Appendix A and listed in Table A2. The total amounts of biogas feedstock, the dry matter content in the feedstock and the corresponding amount of methane produced in the biogas process for each scenario are summarized in Table 4.1.

Table 4.1. Biogas feedstock in the different scenarios

Scenario	Feedstock amounts (t/y) / DM content		Methane production (TJ/y)
	Pig manure	*Ley silage*	
B	5 500 / 8%	-	**3.3**
C1	5 500 / 8%	3 830 / 34%	**14.5**
C2	5 500 / 8%	6 840 / 34%	**23.8**
C3	5 500 / 8%	5 250 / 34%	**18.2**

The biogas plant

Table 4.2 summarizes background data on GHG emissions and Table 4.3 some of the outcomes from the assessment of the biogas plant performed in part one of the project. The data shown in Table 4.3 are used in the farm based case study. The background to each process step and energy input or emission is further explained in the following text.

Table 4.2. GHG emissions for energy carriers or methane emission.

		Energy source	GHG emissions
Electricity	*Base case*	Swedish electricity mix	10.1g CO_2-eqv./MJ[a]
	Sensitivity analysis	Nordic electricity mix	34.9g CO_2-eqv./MJ[b]
Heat	*Base case*	Natural gas	69g CO_2-eqv./MJ[a]
	Sensitivity analysis	Wood chips	2.2g CO_2-eqv./MJ[a]
Methane	*Global warming potential (GWP_{100})*		25g CO_2-eqv/g CH_4

[a] Gode et al., 2011
[b] Nordic/EU-27 average mix (STEM, 2011; IEA, 2012)

Table 4.3. Summary of outcomes from the biogas plant assessment that are further used in the farm based case study.

Biogas production	**Energy input/emission**
Pre-treatment by extrusion	50 MJ/t ley
Electricity (pumping/stirring)	54 MJ/t feedstock
Process heat (digester heating and feedstock hygienization)	121 MJ natural gas/t feedstock
	143 MJ wood chips/t feedstock
Methane leakage in production	0.27% of methane production
Biogas upgrading	
Electricity demand in upgrading	3% of energy in upgraded biogas
Methane leakage in upgrading	1% of total methane production
Methane leakage in flair	0.02% of total methane production
Gas grid injection	
Propane addition	Propane energy corresponding to 25% of the energy in the total methane production[a]
Biogas compression	
Electricity for compression	Electricity corresponding to 2.6% of the energy in the upgraded biogas.

[a]The propane replaces natural gas in the grid, the GHG emission is 4.7g CO_2-eqv./MJ' the difference between life cycle GHG emissions from propane (73.7) and natural gas (69). JRC (2011).

Pre-treatment

Ley can be problematic as biogas feedstock. Even if it has been chopped and ensiled, a floating layer can be formed in storage tanks and reactors. In the biogas process configuration at hand in the present evaluation, all feedstock is mixed to slurry in a reception tank. The slurry is then pumped to hygienization tanks for heating before entering the digester. In Germany, where crop based biogas production is common; this is a less common method. Only 9% of monitored biogas processes used this method, while the majority applies feeding of the crop feedstock directly into the digesters (FNR, 2010). Under these conditions, pre-treatment is assumed to be a prerequisite for enabling adding ley as a feedstock. One of several methods that have been applied for ley pre-treatment in order to avoid the floating layer is extrusion. The extrusion method as suitable for pre-treatment of ley crops for biogas production has been verified at e.g. Foulum biogas plant, Aarhus University, Denmark (Foged et al., 2012). The energy input for extrusion has been given as an electricity demand of 14 kWh/t for ley silage at 25-30% DM, which is assumed valid also for the ley silage with a slightly higher DM content (34% after ensiling losses) in the present study (Lehmann, 2013).

Heat

The biogas plant is operated at a process temperature of 37°C. All feedstock is also heated to 70°C for one hour in order to fulfil the hygienization requirements stated in the EU directive on animal by-products not intended for human consumption (EC, 2002). This is only required for manure in this case, but the process configuration, where all feedstocks are mixed in a reception tank, will make hygienization also necessary for the ley that is added. In 2011, the biogas plant used 6.5 TJ natural gas to produce process heat, corresponding to 6.2 TJ heat with an assumed efficiency of 95%. This corresponds to a natural gas demand of 121 MJ/t feedstock. In the sensitivity analysis, the natural gas for heating is replaced by solid biofuels. With a conversion efficiency of 80% for wood-chips based heat, the demand for wood chips is 7.7 TJ or 143 MJ/t feedstock.

Electricity

The electricity consumption measured for the anaerobic digestion (for pumping, stirring etc.) was 2.7 TJ in 2011, which corresponds to 54 MJ/t of feedstock.

Methane losses

There is a risk of methane leakage from the biogas plant. All the feedstock at the studied biogas plant is handled in a closed building with controlled ventilation, where the methane emissions via the ventilation are checked once a year. For 2011, these measurements indicated a total methane loss of 5 900 m^3, which corresponds to 0.27% of the total methane production. Other potential sources of methane loss from the biogas process are not measured but estimated to be small and thus not included here. When expressed as global warming potential (GWP100), 1 kg of CH_4 corresponds to 25 kg CO_2-eq. (Forster et al., 2007).

Biogas handling

Upgrading

The biogas is upgraded with pressure swing adsorption (PSA) technology. The electricity consumption was measured to be 3.18 TJ in 2011, corresponding to 3% of the amount of energy in the upgraded biogas. The methane slip from the upgrading unit has varied from 0.7-1.4% in different measurements. Here, calculations are based on an average methane slip of 1% of the produced gas.

1% of the biogas produced is not upgraded but flared due to planned and unplanned downtime in the upgrading plant. The efficiency in the conversion of CH_4 into CO_2 in the flare is here estimated to 98% on average. Thus, 2% of the biogas which is flared is not converted into CO_2 but released as CH_4, corresponding to 0.02% of the produced biogas.

Gas grid injection

In 2011, the natural gas distributed via the Swedish gas grid had a heating value of approximately 39.5 MJ/m^3 (lower heating value, LHV) (Swedgas, 2012). When upgraded biogas is injected into the grid, the LHV must correspond to that of the natural gas. Upgraded biogas, with a CH_4 content of approximately 97%, has a LHV of 34.8 MJ/m^3. In order to reach the same heating value as natural gas in the Swedish gas grid, propane (liquid petroleum gas, LPG) with a heating value of 93.4 MJ/m^3 is added before the injection into the gas grid. In 2011, the amount of propane added was 23.1 TJ, corresponding to 25% of the amount of energy in the biogas produced. The LPG added is assumed to replace natural gas and the net increase in GHG emissions is attributed as an emission of the biogas system.

Compression

Upgraded biogas utilized as a vehicle fuel is compressed to 200 bar at the filling station, which requires an input of electricity equivalent to 2.6% of the energy in the upgraded gas (JRC, 2011; Lantz et al., 2009). According to JRC (2011), CH_4 losses from the filling station are insignificant and thus not considered here.

5 CULTIVATION

Here, key features of the calculation methods and background data are summarized. The data used for base case scenarios are described, and if alternative data are used in sensitivity analyses these are presented. For details on cultivation aspects such as fertilization, see Appendix B.

5.1 CROP PRODUCTION

Average crop yields were estimated based on annual measurements on the Wrams Gunnarstorp farm. For cereal grains and rapeseed average production yields were calculated, Table 5.1. Yields for ley crops were estimated from hand-harvested samples and corresponding machinery field losses (20%) within an ongoing research project at SLU, evaluating ley crop yields on the Wrams Gunnarstorp farm (funded by Stiftelsen Lantbruksforskning, SLF). A ley crop system with two harvests per year was assumed.

Table 5.1 Yield data used in the systems analysis and calculation of soil carbon changes.

Crop	Plant part	Yield [t DM/ha]
Winter wheat	Grains	6.5
Ley crops, 0 year after oats	Above-ground	1.5
Ley crops, 0 year after wheat	Above-ground	2.5
Ley crops, 1st of one year	Above-ground	9.0
Ley crops, 1st of two years	Above-ground	12.0
Ley crops, 2nd of two years	Above-ground	9.0
Oats	Grains	4.0
Winter rapeseed	Seeds	2.5

Ley crops were assumed to be sown together with the preceding crop (scenarios A, B, C1 and C2: oats; scenario C3: wheat). After the preceding crop has been harvested, the ley crops grow up. In the year of establishment, ley crops are assumed to be harvested once in the late autumn. Oats are assumed to be harvested 20th of August and ley crops harvested 30th of September are assumed to result in a biomass yield of 1.5 t DM/ha. Winter wheat is assumed to be harvested 10th of August and ley crops harvested 30th of September are assumed to result in a biomass yield of 2.5 t DM/ha. Breaking of the ley crop is assumed to be carried out 1st of August resp. 1st of September in order to allow establishment of a winter crop (rapeseed and wheat, respectively). In this year, the ley crop is assumed to yield 9.0 t DM/ha instead of 12 t DM/ha in a full production year. A full production year is only possible the first year of a two-year ley crop.

5.2 AMOUNTS OF CROP RESIDUES

Crop yields play a central role in this study, since many analysis parameters are directly or indirectly connected to biomass and/or grain yields. Higher crop yields often result in larger amounts of crop residues, e.g. straw, stubble, roots and extra root biomass. Most models for calculation of crop residues assume 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. However, some of these ratios, e.g. for grain/straw, vary strongly and one example is the different developments of straw lengths in the regional breeding programs (Nilsson & Bernesson, 2009).

Swedish studies support models that results in high biomass respective carbon inputs from root and extra root material, especially in ley crops. Ley crop as 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. Another aspect of ley crops is the time factor. High production systems may utilize ley crop blends for 1-3 years, while more long-term or permanent ley crop systems exist as well. The proportion of grasses and legumes within a blend may change over time, e.g. as influenced by (mainly) the level of nitrogen fertilization.

Root biomass in ley 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; Bertilsson 2009). However, in this study, a proportional root biomass development was assumed with a ceiling value of 6 t DM/ha. The impact of these model changes are presented in the sensitivity analysis (Chapter 8).

In the base case calculations, Nordic data is used for the calculation of amounts of crop residues as described above. For crop specific details, see Table E3 and Table E4 in Appendix E. In a sensitivity analysis, IPCC default data are used (Table E1 and Table E4). The nitrogen content in the crop residues is shown in Appendix C, Table C1, and 1% of the nitrogen left in the soil after harvest in crop residues is assumed to generate N_2O (IPCC, 2006). The amounts of crop residues have also an impact on the soil carbon changes, which is described in the following section.

5.3 SOIL CARBON CHANGES

Soil carbon content is a parameter that connects soil fertility, crop yields and emissions of carbon dioxide equivalents and therefore is a crucial factor for the environmental performance of a cropping system. Processes leading to negative changes in carbon soil content are often slow; therefore the time perspective becomes important. While changes in the short-term might be negligible, the long-term effects may lead to substantial changes in soil fertility, crop yields and emissions of CO_2 equivalents. It is therefore also important to look at long-term effects of carbon input on the soil carbon content. IPCC suggests a period of 20 years to be used for this purpose. In theory, changing from one established cropping system to another with a change in the average carbon input may result in a change of soil carbon content. How quickly the change will take place and at what time a new steady state is reached depends on many factors. However, the soil carbon content will change along an asymptotic curve, with the highest absolute changes in the beginning. With time, these differences will become smaller and smaller until a new steady state is reached. Therefore, a shorter period for calculating the average annual carbon change will result in higher changes compared to a longer calculation period. In the base scenario we used a calculation period of 40 years for the Nordic conditions, while 20 years is also investigated in the sensitivity analyses.

Soil carbon content affects many cultivation factors such as nutrient availability, water retention capacity, soil density, soil temperature etc. An increase of soil carbon content may lead to, but is no guaranty for, increased soil productivity, i.e. increased crop yields. But also decreased requirements for fertilization can be a positive result of soil carbon increase.

Soil carbon content is a seemingly stable parameter that can nonetheless change considerably over a longer period of time. The inflow of carbon is restricted to crop residues left in the field and other organic amendments such as manure or digestate. The outflow from the pool of carbon can be simplified as two-fold: a young carbon pool that receives all carbon inflow. From this pool, easily degradable carbon is released as CO_2 according to degradation function, while only a material-specific fraction is humified, i.e. stored in the humus part of soil carbon, or, in this model, the old carbon pool. Outflow from the old carbon pool is generally much lower and decreasing with increasing latitudes due to weather conditions.

Changes in soil carbon content were calculated employing the ICBM calculation model that is well-described in literature (Andrén & Kätterer, 1997; Kätterer & Andrén, 2001). The model was applied to calculate the soil carbon content according to carbon inputs and mineralization rates. For this purpose, the model was modified to account for different input types with specific humification factors. Prior to analysis of the project scenarios, the model was calibrated with data derived from long term soil carbon field trial located in proximity to the studied farm. For details on the model, its parameterization and calibration see Appendix E, Chapters E1, E3 and E4 respectively.

5.4 REPLACING MINERAL FERTILIZERS

In all scenarios, biofertilizers are used. The amounts of biofertilizers and nutrient concentrations after storage are shown in Table A5 in Appendix A. Nutrient amounts for the different scenarios after storage losses (at field application) are summarized per ha for each scenario in Table 5.2. The nutrient content in the biogas feedstocks and digestate, and the calculations for conversions occurring in the biogas process are described in Appendix A. After losses at field application, as described in Appendix C, the NH_4-N in the digestate is assumed to replace mineral nitrogen. For P and K, no losses are subtracted. The emissions related to mineral fertilizers production are described in Appendix B. For Scenarios A and B the added nutrient amount after subtracted losses are equal. For the C-scenarios, nutrient addition after losses varies due to differences in nutrient demand in the different crop rotations.

Table 5.2. Nutrient demand and supply by biofertilizer or as mineral fertilizers in the different scenarios as average per ha for the crop rotation on 650 ha.

	Total demand based on mineral fertilizer application			Added as biofertilizer (manure (A) or digestate (B and C) (kg/ha)				Added as mineral fertilizer (kg/ha)		
Scenario	**N**	**P**	**K**	**N-tot** [b]	**NH4-N** [b]	**P** [a]	**K** [a]	**N** [b]	**P** [a]	**K** [a]
A	196	28	42	32	21	10	15	173	18	27
B	196	28	42	33	27	10	15	169	18	27
C1	183	27	76	87	55	16	47	134	11	28
C2	179	28	103	120	69	20	67	120	8	36
C3	192	28	87	104	62	18	57	138	10	30

[a] For P and K the losses after field application are assumed to be no different for mineral fertilizer and biofertilizers

[b] For N, the loss as NH_3 after application is 0.9% of added N for mineral fertilizer. For the biofertilizers, 15% of added N is assumed to be lost as NH_3 when applied in cereals, 30% when applied in ley.

Note that the reduction in mineral fertilizer demand shown above when ley is introduced in the C-scenarios only includes the reduction made possible by recirculation of N, P and K in the ley based digestate to the fields. In addition, introduction of ley will give a preceding crop effect that has not been taken into account in the present calculations.

5.5 BIOGENIC NITROUS OXIDE EMISSIONS

Nitrogen related emissions and the background to calculations methods and emissions factors are described in Appendix C. In cultivation, both nitrogen in crop residues and added N fertilizer contribute to biogenic N_2O emissions. In the base case, the IPCC default emissions factor of 1% is used for all N additions except for mineral fertilizer and manure, where national emissions factors of 0.8% and 2.5% are used (IPCC, 2006; Naturvårdsverket, 2013). In the sensitivity analysis, the IPCC default value is applied also for manure. In an additional sensitivity analysis, Swedish experimental emissions factors are applied for both manure (0.9%) and digestate (0.2%) (Rodhe et al., 2012; Rodhe et al., 2013). In addition, amounts of crop residues based on different calculation models are varied in the sensitivity analysis.

5.6 SOIL COMPACTION

Soil compaction is a general problem in crop cultivation, especially on heavy clay soils. It is caused by the heavy equipment used for tillage, biofertilizer spreading, combining and other field operations and it leads to reduced crop yields. An estimation is that the grain yield, on average, has been reduced by 15-25% on the farm in question (where soils have a clay content of up to 65%) due to soil compaction in combination with a low soil carbon content (Rasmusen, 2008). There is a risk of increased soil compaction when digestate replaces mineral fertilizers since the machinery and the digestate together will be significantly heavier, or about 50-60 t in systems using tractor and liquid manure spreader, than the equipment for spreading mineral fertilizer, or about 10 t using tractor and mineral fertilizer spreader. Based on a soil compaction model developed at SLU (Arvidsson, 2008), the soil compaction on fields like the ones included in this assessment is found to increase when using a tractor and a liquid manure spreader, resulting in approximately 20% lower crop yields in the long-term. Critical parameters in soil compaction are clay content, soil moisture content, the weight of the field equipment, wheel load, the size of tires, tire inflation pressure etc. (Arvidsson, 2008; Arvidsson & Håkansson, 1991).

At the biogas plant analysed in part one of this project, an alternative technology for spreading the digestate is however used for 48% of the digestate (Lantz & Börjesson, 2013). This technology is the umbilical, slurry spreading system. This system includes pumping the digestate from the storage tanks via a mobile pipeline to the surrounding fields where the digestate is spread using equipment with a hose reeler unit (a drag hose), all mounted on a self-propelled machine (manufactured by Agrometer, Denmark). The weight of this field equipment is, on average, 12 t, thus only slightly more than the weight of the mineral fertilizer spreader. The resulting minor increase in soil compaction is calculated to reduce crop yields by only 1.5% (Arvidsson, 2008). At the farm analysed in the case study, all the 650 ha are within close distance from the digestate storage tanks and the umbilical, slurry spreading system is used for digestate application on the whole area. Since a crop yield decrease of 1.5% lies within the uncertainty of the calculation method, no compaction effect of the digestate spreading compared to mineral fertilizer spreading was assumed in the farm based case study.

5.7 SYSTEMS EXPANSION

In the systems expansion, the reduced food crop production when ley is introduced in the C-scenarios was assumed to be compensated for by additional grain and oil seed production outside the farm but within the region on excess farmland. Thus, the total output of wheat, oats and rape seed is equivalent in all investigated scenarios. The data used for the GHG emissions in cultivation are taken from Börjesson et al. (2010). The given values are recalculated per kg DM of crop grain or seed and shown in Table 5.3.

Table 5.3. Greenhouse gas (GHG) emissions for crop production included by systems expansion.

Systems expansion: emissions from replacing crops		
Oats	407	kg CO_2/t DM
Rape	829	kg CO_2/t DM
Wheat	407	kg CO_2/t DM

6 STORAGE OF MANURE AND DIGESTATE

6.1 BIOGENIC NITROUS OXIDE EMISSIONS

Nitrogen-related emissions and the background to the calculation methods and emission factors are described in Appendix C. In Scenario A, manure is assumed to be stored under a floating crust cover, while digestate is assumed to be stored under a permanent roof cover (as is the case at the full scale biogas plant assessed in part one of the project). The nitrous oxide (N_2O) emission is under these conditions 0.5% of N-tot content for manure, and for digestate assumed to be zero (Appendix C). In the large scale process investigated in part one of this project, the very low dry matter content of the digestate would likely prevent crust formation. Even with the high fraction of ley addition investigated in Scenario C2, the DM content of the digestate would be below 5% with the present waste mix fed to the co-digestion plant. A case with crust cover on the digestate storage is thus not relevant for the studied scenarios. It could, however, be relevant in other biogas processes, or if the waste mix would include a higher fraction of high-DM feedstocks.

6.2 METHANE EMISSIONS

CH_4 emissions and the background to calculations methods and CH_4 conversion factors (MCF) are described in Appendix D. The default value for MCF used in the Swedish national inventory report, 3.5%, is in the base case used for both manure and digestate storage (Naturvårdsverket, 2013). In the sensitivity analysis, alternative MCF values based on Swedish experimental data is used; 2.8% for manure and 12% for digestate (Rodhe et al., 2012; Rodhe et al., 2013). In a second sensitivity analysis based on total methane production, the leakage in digestate storage is calculated to correspond to 4% of the produced and collected methane in the biogas process.

7 RESULTS AND DISCUSSION

7.1 GREENHOUSE GAS EMISSIONS

The greenhouse gas (GHG) emissions for each scenario given as CO_2-eq/ha, yr. are shown in Table 7.1, and illustrated graphically in Figure 7.1. Negative values indicate avoided emissions.

Table 7.1. Base case greenhouse gas (GHG) emissions per category for the five investigated scenarios

GHG emissions (kg CO_2-eq/ha, yr)	Scenario				
	A	B	C1	C2	C3
Cultivation diesel	230	229	275	288	304
Cultivation materials/machinery	292	291	284	260	287
Cultivation fertilizer production	1 216	1 179	891	864	891
Biogenic N_2O mineral fertilizer	650	632	502	451	515
Biogenic N_2O biofertilizer	373	155	409	563	489
Biogenic N_2O crop residues	179	179	298	320	349
Biogenic N_2O indirect	170	175	203	196	212
Biogas process and pre-treatment energy	-	75	131	174	151
Biogas upgrading energy and propane	-	9	39	63	49
Biogas process methane leakage	-	7	33	52	41
Biogas upgrading methane leakage	-	25	112	180	141
Emissions manure/digestate storage	229	57	91	136	102
Soil carbon change	-869	-803	-2 128	-2 425	-2 592
Soil organic matter N uptake	-111	-103	-272	-310	-331
Net emission	2 359	2 107	868	812	608

The evaluation of the effect of introduced biogas production is done in two steps. The first change is the production of biogas from pig manure, a change that has already been made at the investigated farm. Here, Scenario A is the reference for Scenario B. The next change to be introduced is ley production on parts of the farm land, in order to complement the manure as a raw material for biogas production. This change is evaluated by comparing the C scenarios with scenario B as reference scenario, subtracting emissions from scenario B from all the scenario C emissions. In Figure 7.2, the resulting GHG emissions after subtracting the reference scenario emissions are shown grouped into categories.

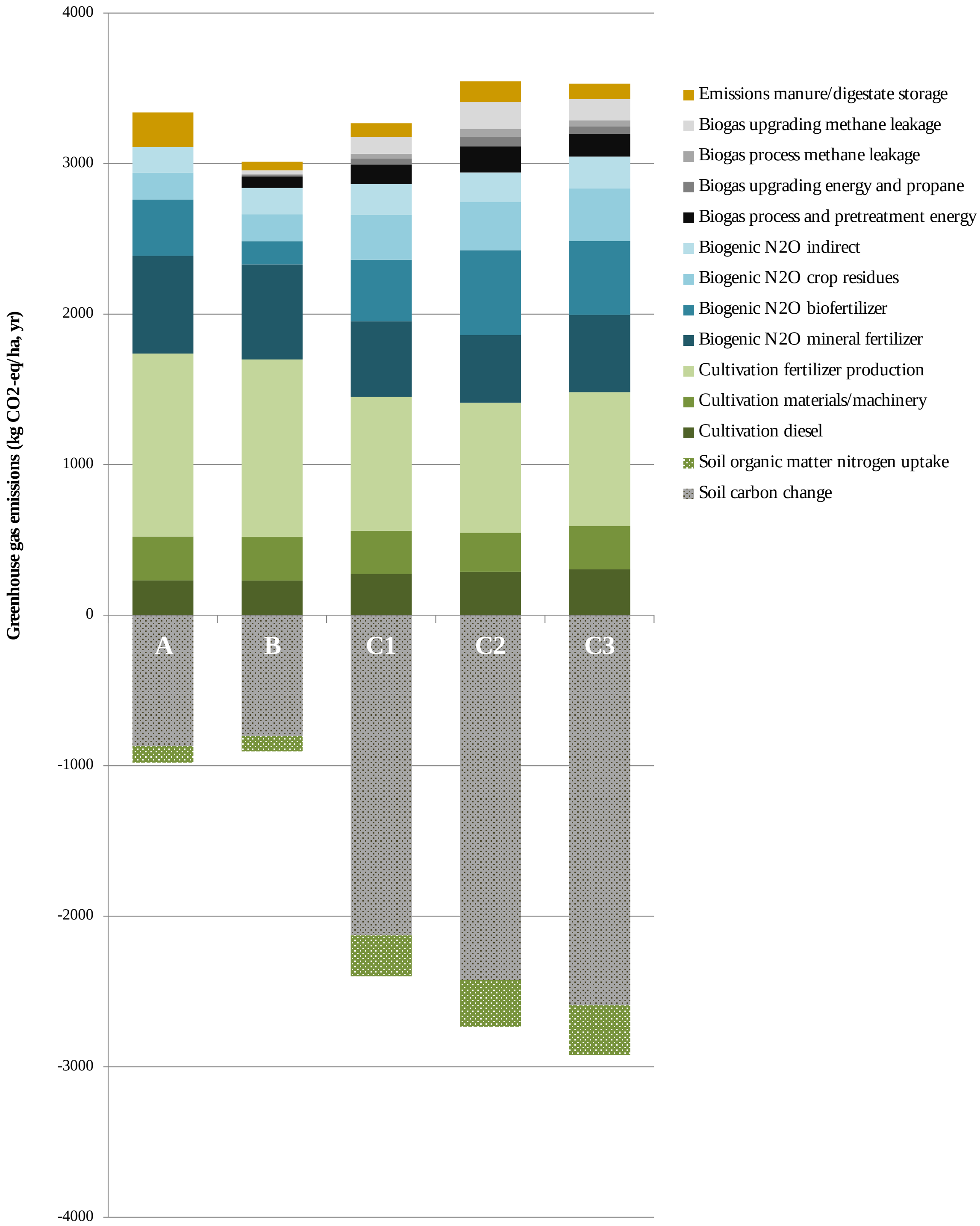


Figure 7.1. GHG emissions for the different scenarios per hectare and year. The categories are organized in colour themes that are reflected also in Figure 7.2. Soil organic matter changes: dotted. Cultivation except biogenic N_2O emissions: green. Biogenic N_2O emissions: blue. Biogas production: grey/black. Manure or digestate storage: yellow. Negative values indicate avoided emissions.

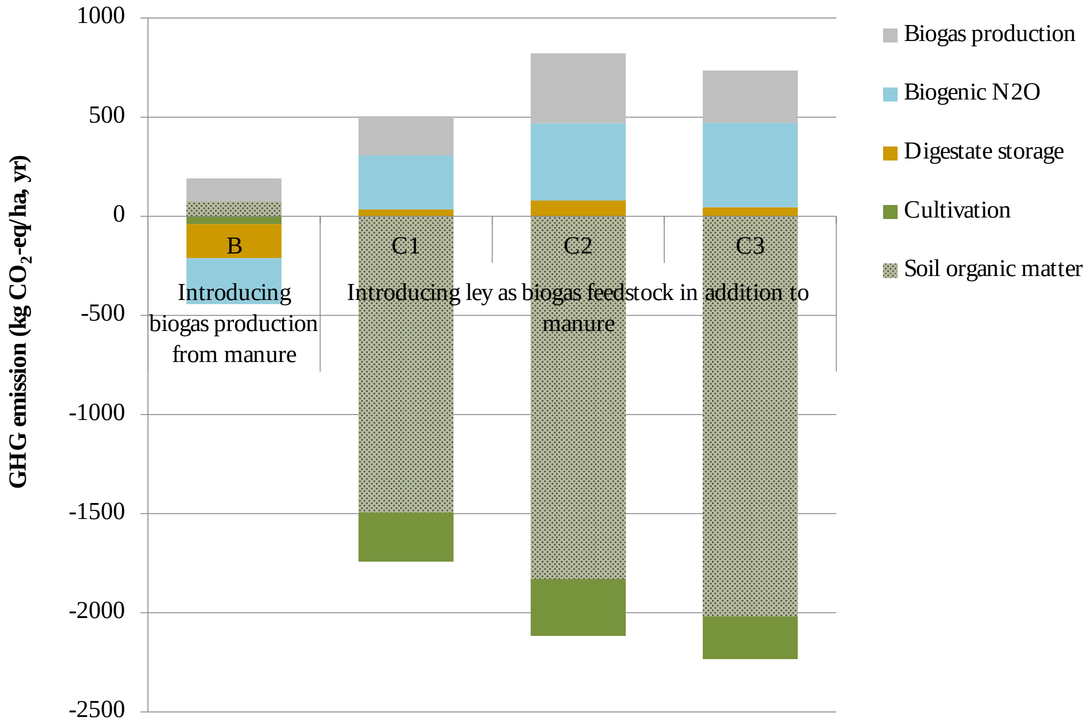


Figure 7.2. GHG emissions per hectare and year for the changes studied grouped into categories. Negative values indicate avoided emissions.

As shown in Figure 7.1, in a conventional farming scenario (A) the main source of GHG emissions is the production of mineral fertilizers even though already here some pig manure is used as biofertilizer, which reduces the need for mineral fertilizer (Table 5.2). Another large contributor is the biogenic N_2O emissions from the soil. These emissions are proportional to the addition of nitrogen fertilizer, but also to the nitrogen that is left in the field as crop residues. In scenario A, the pig manure used as biofertilizer adds on average 32 kg/ha of total nitrogen (N-tot), whereof 21 kg/ha is in the form of ammonia nitrogen (NH_4-N) (Table 5.2). The calculation of N_2O emissions is based on the N-tot addition, so organically bound nitrogen, which is not included as available for the crop when total N-fertilization requirements are to be met, will here only contribute negatively to emissions. When the manure is digested in scenario B, less organically bound nitrogen is added due to the conversion to NH_4-N. This will also reduce both the emissions from production and addition of mineral fertilizer.

An important contribution to the reduced emissions between scenarios A and B is the fact that the digested manure is assumed to emit less N_2O than non-digested manure due to the removal of easily degradable carbon in the digestion process (see Appendix C). This removal of carbon through the production of biogas will also have an impact on the build-up of soil organic matter. Through the anaerobic digestion process, the easily degradable part of the organic matter in the manure is converted to CO_2, CH_4 and microbial biomass. Through this process, 55% of the carbon in the pig manure is lost, corresponding to 134 kg C/ha, yr. on average for the farm, while another 3% is assimilated into microbial biomass, which stays in the digestate. It is, however, the fraction of organic material that is less readily biodegraded that remains in the digestate, and a higher share of this organic matter forms stable soil organic matter in the digestate than in the non-digested manure. The impact on long term stable soil carbon content is thus small in comparison to the carbon removal from the system. The biogas production from the manure gives a decrease in contribution to long term stable soil carbon with on average 18 kg C/ha, yr. for the farm. The effect of this on long term soil organic matter build-up is further discussed in Chapter 7.2. A third aspect related to the removal of organic matter through biogas production is the reduced leakage of GHG from digestate storage compared to manure storage. This is partly due to reduced methane leakage, but also due to the fact that the naturally crust covered open manure storage emits both N_2O and NH_3, while the emissions from the roof covered digestate storage are much lower.

In the C-scenarios, the emissions related to diesel, machinery and materials (described in Appendix B) in cultivations change little compared to in the B scenario. The emissions related to both mineral fertilizer production and use, however, decrease much when the amount of available biofertilizer increases. The biogenic N_2O emission from the addition of the biofertilizer and from the increased amounts of crops residues originating from ley cultivation do at the same time increase, and the total GHG contributions from cultivation inputs and biogenic N_2O emissions is similar to or higher for the C-scenarios than for the reference B scenario (green + blue bars, Figure 7.2). The emissions related to the biogas production are also higher for all the C-scenarios, where more biomass is processed and more biogas is produced, than for the B scenario where only manure is used for biogas production. The large benefit of the C-scenarios is, however, the major contribution to build up of soil organic matter that occurs. This is both an effect of increasing amounts of crop residues when ley is introduced in the crop rotations, but also the effect of the increased amount of biofertilizer available in form of digestate. Of the carbon in the ensiled ley used for biogas production, 51-53% is removed through CO_2 and CH_4 in the biogas produced. The rest remains in the biofertilizer, and is together with the nutrients recirculated to the farm land. The average carbon addition through digestate is 539, 876 and 718 kg C/ha for C1, C2 and C3 respectively. This higher addition of biofertilizer gives a drawback in increased N_2O emissions (Figure 7.1), but the total benefit on GHG emissions of digestate addition is much greater than the drawbacks. The benefit of soil organic matter indicated in Figure 7.2 includes both the carbon sequestration and the integration of nitrogen in the soil organic matter, giving a reduced biogenic N_2O formation.

Systems expansion

In addition to the GHG emissions generated or avoided per hectare, each scenario produces dried grains/seeds and upgraded and compressed biogas delivered to the gas grid. The products generated per scenario are shown in Figure 7.3. The yield of crops per ha is then the same for all scenarios, and the potential for changes in crop yield due to the changes in soil organic matter is later discussed in section 7.3. The decrease in food crop production or increase in biogas production after the introduced changes, are shown in Figure 7.4. The loss of food crops or gain of biogas is integrated in the assessment as corresponding GHG emissions in a systems expansion. The biogas produced in Scenarios B and C is assumed to replace fossil vehicle fuels, using the default GHG emission value in the EU renewable energy directive of 83.8 g CO_2-eq/MJ (EU, 2009). The end use is included as recommended in the directive, i.e. without taking into account potential end use emissions when biogas is used to replace petrol and diesel, or any difference in fuels use efficiency (EU, 2009). The GHG emissions from biogas end use in the current gas vehicle fleet in Sweden have also been shown to be low, having little impact on the final result (Tufvesson et al., 2013). The fuel use efficiency could however potentially be an issue when replacing diesel use for heavy vehicles.

The lost crop production is recalculated to GHG emissions based on emissions that would occur when cultivating these crops elsewhere in the region. It is assumed that excess crop land is available outside the farm, where additional grain and oil seed production can take place without any iLUC effects. Emissions have been shown to be 407g CO_2-eq/kg DM for wheat and 829g CO_2-eq/kg DM for oilseed rape (Börjesson et al., 2010). Calculations are based on Swedish regional conditions, and include biogenic N_2O emission, but not soil carbon changes. The study by Börjesson et al. (2010) does not include values for production of oats, but emissions are set as for wheat based on the findings by Ahlgren et al (2011), where GHG emissions from crop cultivation are calculated based on methodology in the EU renewable energy directive (EU, 2009). The resulting effect on GHG emissions of the system expansion, replacing or utilizing the products shown in Figure 7.4, are shown in Figure 7.5.

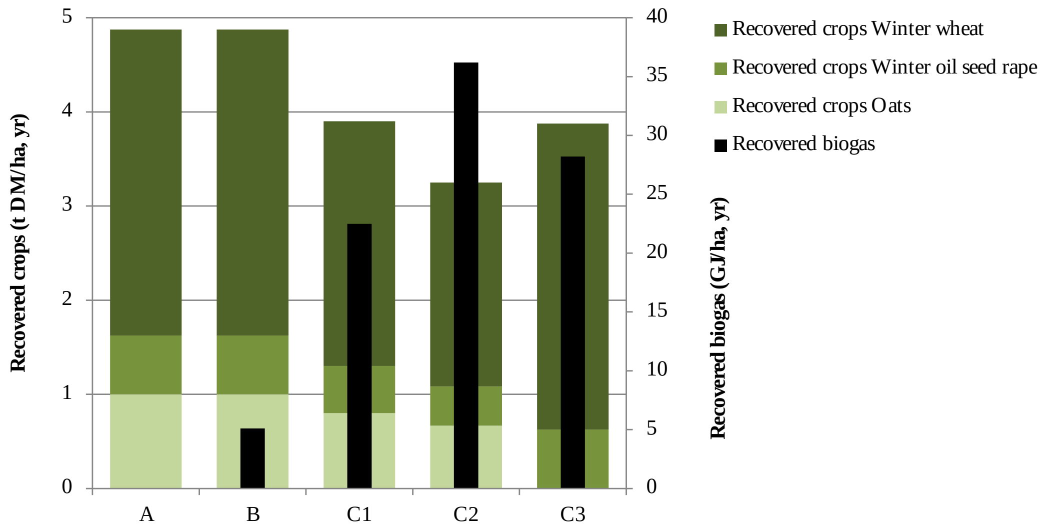


Figure 7.3. Crops and biogas recovered in each scenario as average per ha for the 650 ha farm.

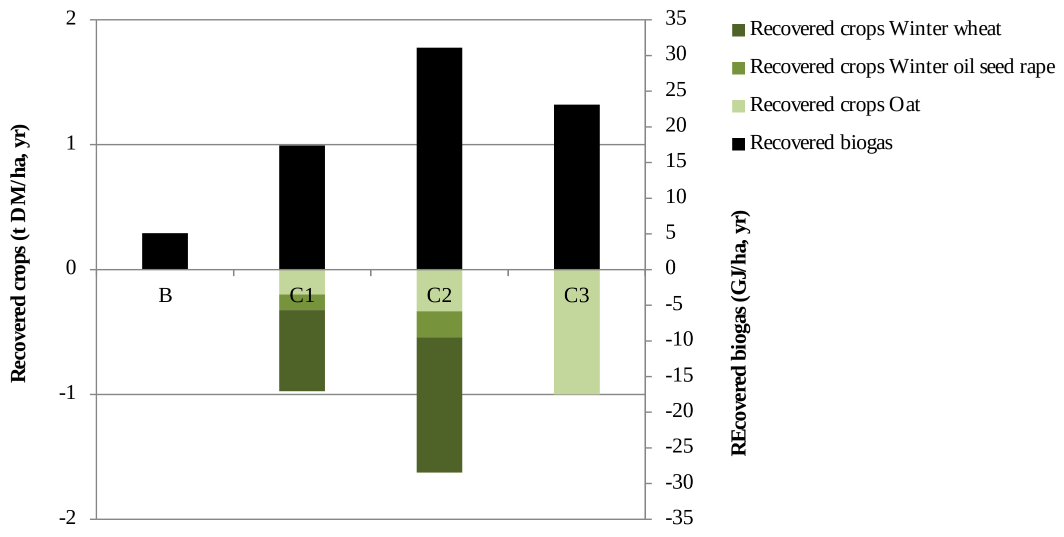


Figure 7.4. Crop production lost (negative values) and biogas recovered as effect of the introduced changes as average per ha for the 650 ha farm.

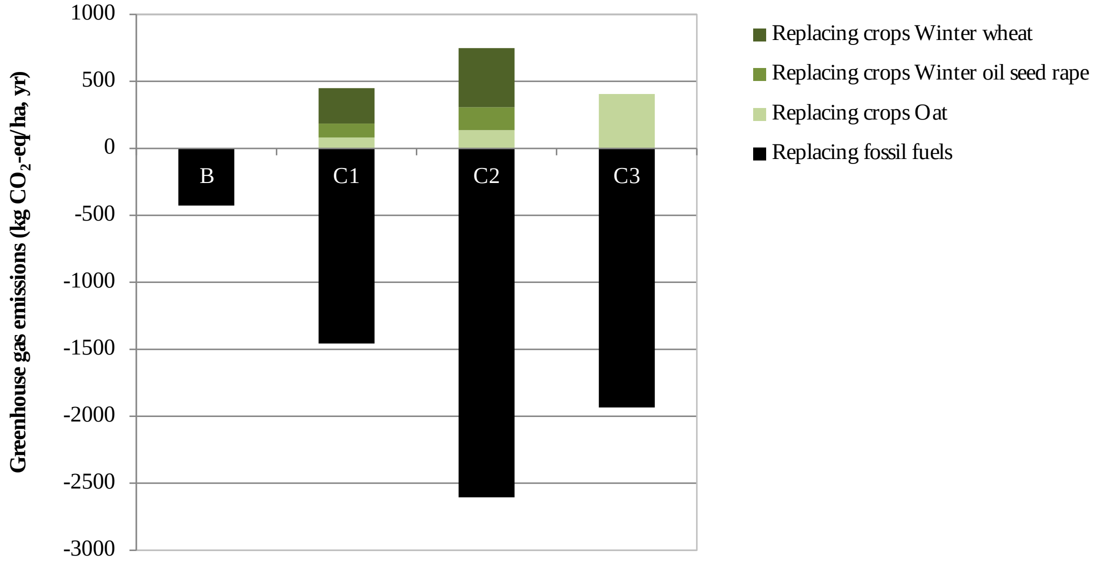


Figure 7.5. Systems expansion. Greenhouse gas (GHG) emissions for replacing lost crop production, and emission savings (negative values) when recovered biogas replaces fossil fuels as average per ha for the 650 ha farm.

The impact on each scenario of system expansion is shown in Figure 7.6. Compensating for the decreased food crop production decreases the climate benefit of the systems. Despite that, the resulting reduction in GHG emission is large. The resulting net GHG emissions after systems expansion are presented in Table 7.2 from the farm perspective, per hectare and year, as average for the 650 ha farm. The emissions are also calculated as avoided emission per GJ fuel produced and utilized for replacing fossil vehicle fuels, to enable comparison with GHG emissions for other vehicle fuels.

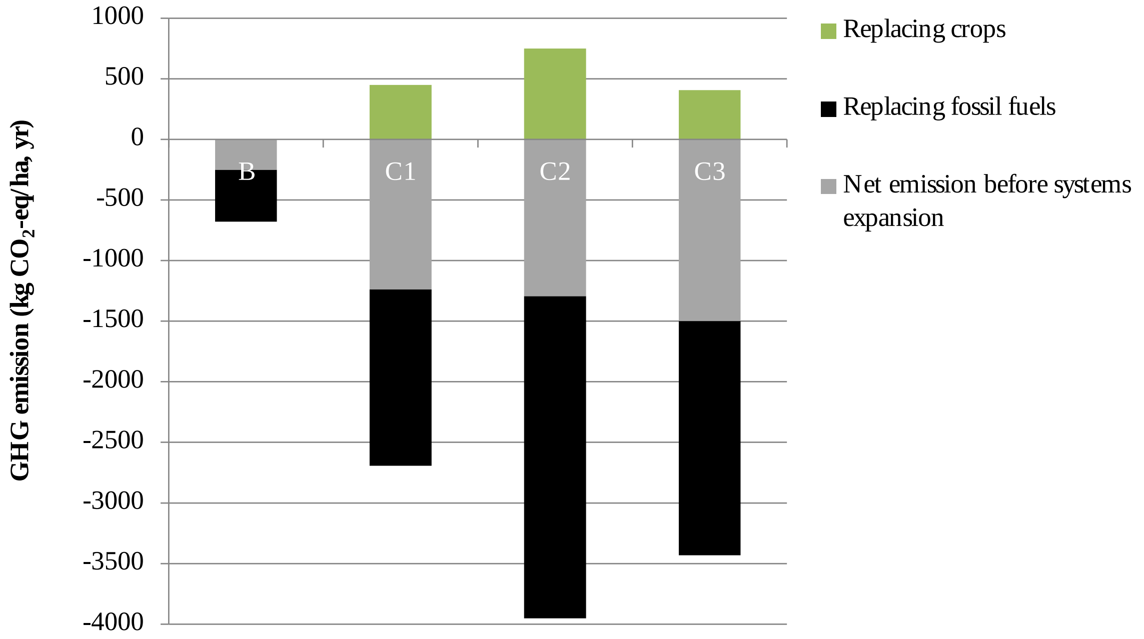


Figure 7.6. Net GHG emissions before systems expansion and the impact on GHG emissions of systems expansion, replacing crop production, and emission savings when recovered biogas replaces fossil fuels as average per ha for the 650 ha farm.

Table 7.2. Net emissions of GHG for the studied changes after systems expansion, where negative values indicate avoided emissions. Emissions are shown as calculated both per ha and year, and per GJ fuel produced.

Scenario	Emission kg CO_2-eq/ha, yr	kg CO_2-eq/GJ
B	-679	-133
C1	-2 244	-100
C2	-3 203	-88
C3	-3 025	-107

The reference GHG emissions for fossil fuels in the renewable energy directive are 83.8 kg CO_2-eq/GJ. Including the end use as replacing fossil fuels, the total emission reduction for the biogas produced in scenario B is 159%. The effect of introducing ley for biogas production at a farm will give a biofuel with an emission reduction of 119% (C1), 106% (C2) and 128% (C3). This can be compared to the emission reduction of 90% presented in part one of this project (Chapter 4), achieved when biogas is produced from a mix of manure and industrial residues and replaces fossil fuels (Lantz & Börjesson, 2013). Emission reductions above 100% show that the production itself, not only the utilization of the biofuel to replace fossil fuels, gives avoided emissions. For scenario B, the avoided emissions of CH_4 and N_2O during storage and after soil application when the manure is handled as digestate are the main causes for the avoided GHG emissions from biogas production (Figure 7.2). When introducing ley in a cereal based crop rotation, the main cause of the avoided emissions from production is the soil carbon build up, both from the crop residues from ley in the crop rotation, and from the carbon-rich digestate that is recirculated at the farm (Figure 7.2). The results are in line with previous results, where a study on a mix of liquid and solid manure for biogas production has given total emissions reductions of between 185-186%, including biofuel end use, (Tufvesson et al., 2013). The higher reduction shown by Tufvesson et al. (2013) partly depends on a greater climate benefit of including solid manure fractions, but also due to a biogas process with less GHG emissions than the process used in the present project. For biogas from ley analysed from a biofuel production perspective, emission reductions of 101-117% have been presented under conditions in Skåne in previous studies (Björnsson, 2013; Börjesson et al., 2013).

A benefit of this farm based case study is the approach of presenting the result on farm level, which shows how efficient a change in a conventional crop rotation is on GHG emission reductions. The GHG reduction per hectare of land is high when ley is replacing food/feed crops, even after including the GHG emissions from crop production elsewhere in the systems expansion. In Figure 7.7, the average reduction of GHG emissions per hectare for the 650 ha farm is shown in relation to the share of land used for ley production. In scenarios C1 and C3, one year of ley is introduced, and the total loss of crops is the same range, 1 t DM/ha (Figure 7.4) on average over the 650 ha. But while C1 is calculated with a proportional decrease of the crops produced in scenario B, C3 involves replacing the relatively low yielding oats with ley crops. The average climate benefit per hectare of the latter is substantially higher (Table 7.2). Introducing 2 years of ley in a 6-year crop rotation as in C2 in comparison to 1 year as in C1 increases the average loss of crops from 1 to 1.6 t DM/ha, and the climate benefit per ha increases with a similar proportion, a factor of 1.4.

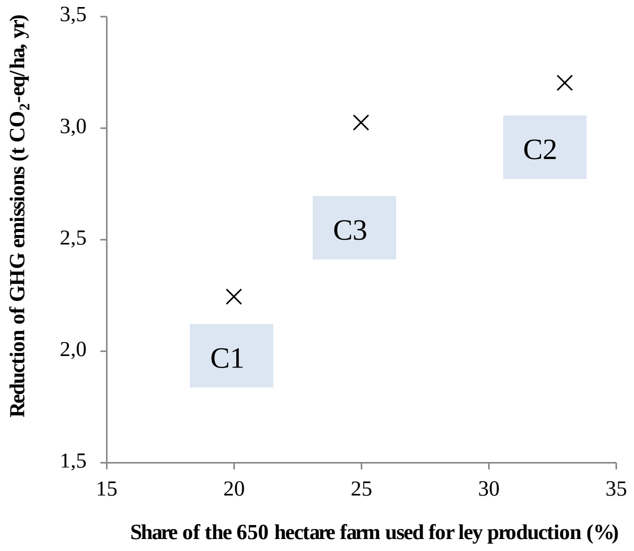


Figure 7.7. Average reduction in greenhouse gas emissions per hectare of the 650 ha farm compared to the share of the farm land taken into use for ley production.

7.2 SOIL CARBON

In the previous section, a build-up or loss of soil carbon was presented based on the climate benefit. In addition, build up- or avoiding losses of soil organic matter is a major issue for preserving soil fertility, which is a highly relevant aspect to consider in relation to land use for energy crop production. Figure 7.8 shows the soil carbon development as calculated for a 50-year period for the five scenarios investigated here, starting at the soil carbon content of 2% in all scenarios, which is the estimated present content at the 650 ha farm.

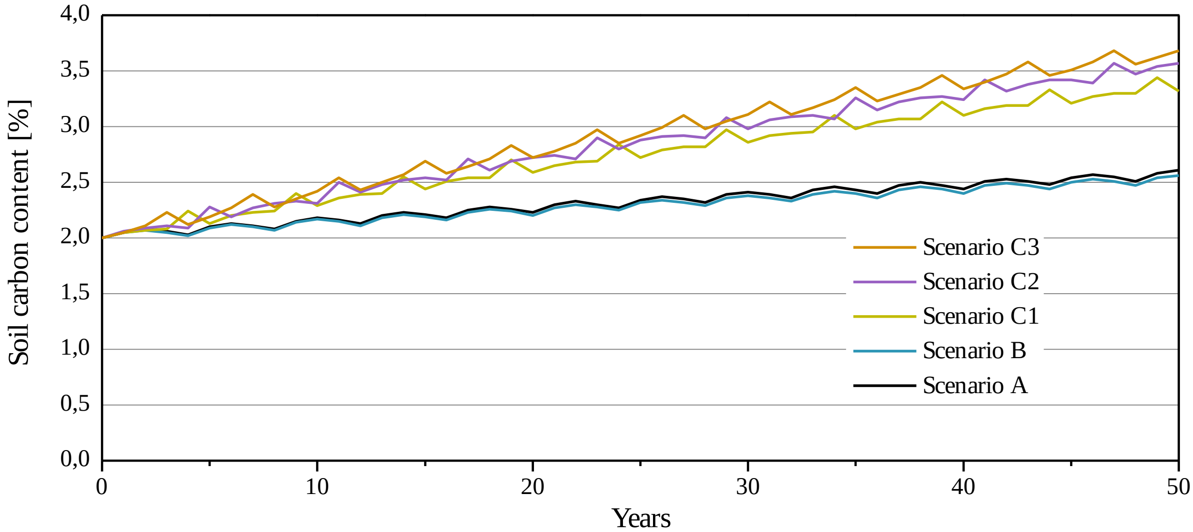


Figure 7.8. Change in soil carbon content in the investigated scenarios over a 50-year period.

The soil carbon content develops positively for all scenarios, but with largely different long-term results. Scenarios A and B have the conventional cereal based 4-year crop rotation, where most of the straw is left in the field and pig manure or digested pig manure is used to cover part of the need for fertilization. In scenario B, where pig manure is used for biogas production before being recycled to farm land, soil carbon content will be slightly lower than in scenario A. However, the difference is small; after approximately 145 and 150 years a new steady state for the soil carbon content is reached at approximately 3.0 and 2.9% for scenarios A and B, respectively. In the C-scenarios, ley is introduced in the crop rotation, and the amount of digestate produced from manure and ley digestion is large enough to cover a large share of the fertilizer need at the farm (Table 5.2). Scenarios C1-3 contribute to a much higher increase in soil carbon content with steady states at 4.1, 4.5 and 5.0% after approximately 125, 130 and 150 years for scenarios C1, C2 and C3, respectively.

7.3 FOOD AND FEED PRODUCTION

Swedish long-term trials have shown that soil compaction can cause crop yield losses of up to 20%, and the higher the clay content, the larger the loss (Arvidsson & Håkansson, 1991). Swedish long-term field trials with and without ley in the crop rotation have shown that the positive effect of 2 years of perennial clover/grass ley in a 6 year crop rotation could give increased winter wheat yields. This increase in yield would then compensate for the loss of land for food production when ley is introduced (Persson et al., 2008). Thus, it is suggested that food crop production could be maintained under certain conditions, while the inclusion of ley at the same time could be a tool to mitigate GHG emissions by soil carbon increases.

In the farm based case study, this mitigation of GHG emissions is further enhanced by

1. the use of ley for production of a fuel that can replace fossil fuels
2. the recirculation of the carbon and nitrogen rich digestate originating from the ley

It would thus be interesting if it was possible to show increased crop yields in the farm based case study, which was the original intention. At the evaluated farm, the low input of carbon in the cereal based crop rotation has been identified as problematic with regards to the high clay content in the present soils. Problems with soil compaction and crop losses due to standing water have been common. The initiative to construct a biogas plant within the farm boundaries was partly due to the wish of obtaining high amounts of biofertilizers to be used in the cereal based crop rotation. Since 2007, all land within the farm is fertilized with the maximum allowed amount of biofertilizer (the content of P is limiting the allowed addition). At the same time, reduced tillage was introduced, where ploughing was substituted by subsoiling. In an estimation in 2008, the grain yield was believed to have decreased by 15-25% on the farm in question (where soils have a clay content of up to 65%) due to soil compaction in combination with a low soil carbon content (Rasmusen, 2008). Since then, when digestate fertilization and reduced tillage was introduced, crop yields show a tendency of increasing, winter wheat yields have increased by approximately 12%, and oats and oilseed rape yields by approximately 20% (Rasmusen, 2013). However, available data is too weak to draw definite conclusions, and the cause is not only the increased amounts of biofertilizer addition. In the present evaluation it was thus decided not to use crop yield increases as linked to increased soil carbon content as a part of the evaluation, but instead include lost food/feed crop production through systems expansion as in section 7.1 and discuss the potential of crop yield increases qualitatively.

Within the EU project Soilservice, the potential yield increase for winter wheat in response to increased soil carbon content under Swedish conditions has been modelled (Soilservice, 2012). In the present evaluation, the increased addition of soil carbon through integration of ley in the crop rotation and the recirculation of biofertilizer would increase stable soil carbon content from 2% to 3%

within 20-30 years (Figure 7.8). The long term increase would be to levels of 4-5%. At high nitrogen fertilization levels and an increase in soil C from 2% to 3%, the yield of winter wheat has been shown to increase by 10% (Soilservice, 2012). A 10% crop yield increase would decrease the crop losses or give increased production of some crops as shown in Figure 7.9 (left), while the right graph shows the effect of a 20% yield increase. If the increase in soil carbon content would increase crop yields of 20%, one year of ley could be introduced in the crop rotation, as in scenarios C1 and C3, with close to maintained crop production. Crop yield increases will in addition give increased amounts of crop residues, which will further improve soil carbon content.

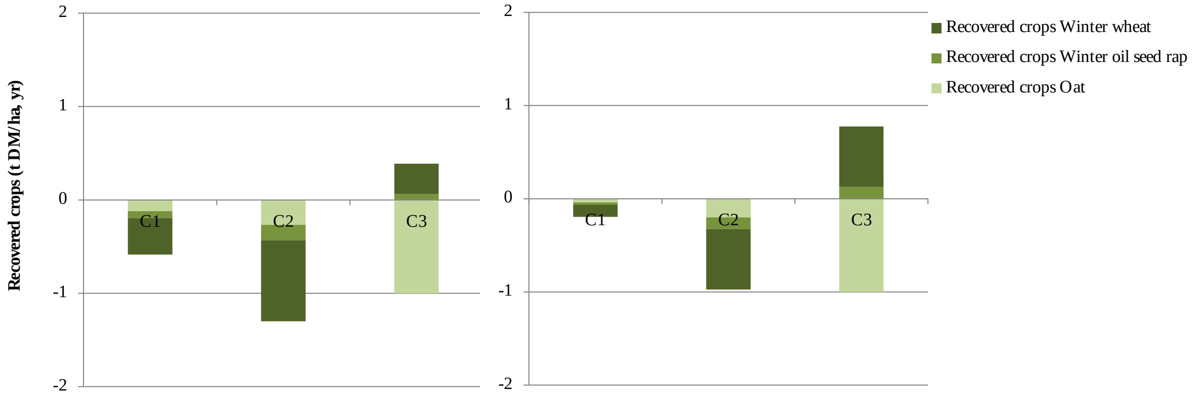


Figure 7.9. Effect of increase crop yields by 10% (left) and 20% (right) on crop recovery in scenarios C1-C3 after introduction of ley on 20%, 33% and 25% of the farm area.

8 SENSITIVITY ANALYSIS

8.1 DIFFERENCE IN CALCULATION METHOD FOR CROP RESIDUES

Two different datasets and methodologies for the calculation of crop residues have been evaluated; IPCC data and Nordic data. The latter is a dataset that has been developed from Nordic literature data within this study. The IPCC method uses a linear correlation with slope and intercept to calculate the above-ground residues from the yields of harvested biomass for most crops (i.e. grains, seeds, tubers etc.), except ley crops, where only a slope is used (IPCC, 2006). In comparison, in the Nordic approach only slopes are used for all crops.

While the global IPCC models assume high shares of straw in the total amount of biomass of cereals, Nordic data points towards shorter and shorter straw lengths (Nilsson & Bernesson, 2009). The IPCC method scales up root biomass proportionally to above-ground biomass. Accordingly, the high above-ground biomass yields used in this study result in very high annual input rates for root biomass in ley crops. IPCC uses a factor for calculating the below-ground biomass yield from the above-ground biomass yield. Similarly, the Nordic approach uses a factor for this, but uses a second factor to calculate extra root material from the root biomass yield. For most crops, these two factors could be combined in a single factor. However, for perennial crops, such as ley crops, the two factors allow calculation of extra root biomass even when the root biomass is not to be accountted for, i.e. in the years the perennial crops are continued. Furthermore, Nordic data limits the amount of root biomass yield to a maximum 6 t DM/ha based on recommendations (Bertilsson, 2009; Bertilsson, 2006). Below this amount, the root biomass yield was assumed to be proportional to the aboveground biomass yield. IPCC does not limit the amount of root biomass for ley crops.

The base case in the present study is calculation of crop residues based on Nordic data, with the described limitation on root biomass amount. Examples on the correlation between above ground harvest of biomass and below ground crop residues based on this method in comparison with the IPCC method is shown in Figure 8.1.

The effect of different input data on amounts of crop residues have impact on the biogenic N_2O-emissions originating from crop residues. The resulting N_2O-emissions are shown in Table 8.1 expressed as CO_2-eq/ha, yr. per scenario. The effects of amounts of crop residues on soil carbon calculations are further discussed in the following chapter.

Table 8.1. Biogenic N_2O emissions from crop residues expressed as CO_2-eq/ha, yr. Base case (bold) and results of sensitivity analysis on input data for yield of crop residues.

	Scenario				
Calculation method	A	B	C1	C2	C3
Base case – Nordic limited	**179**	**179**	**298**	**320**	**349**
Nordic unlimited	179	179	456	514	545
IPCC	323	323	458	379	539

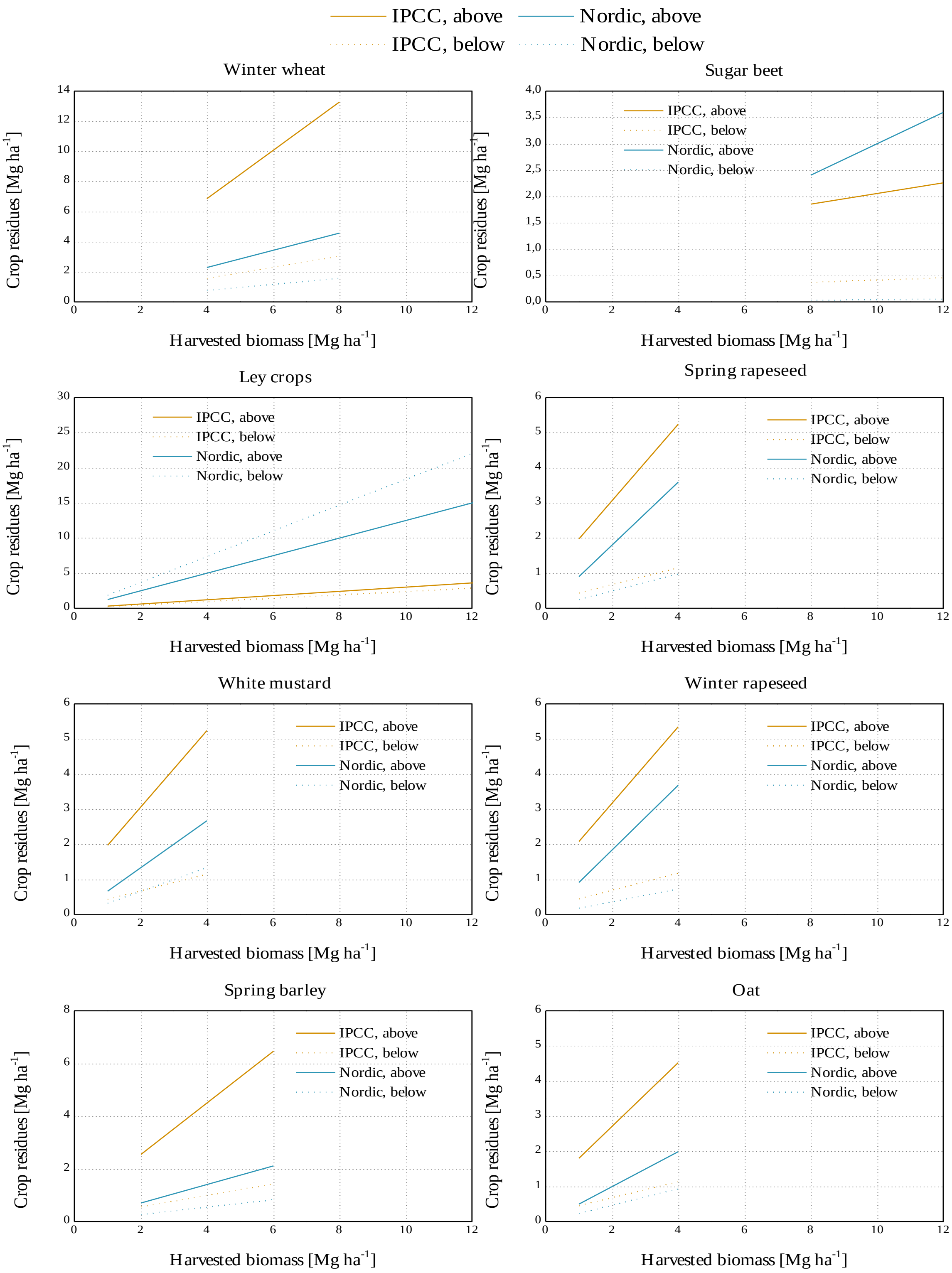
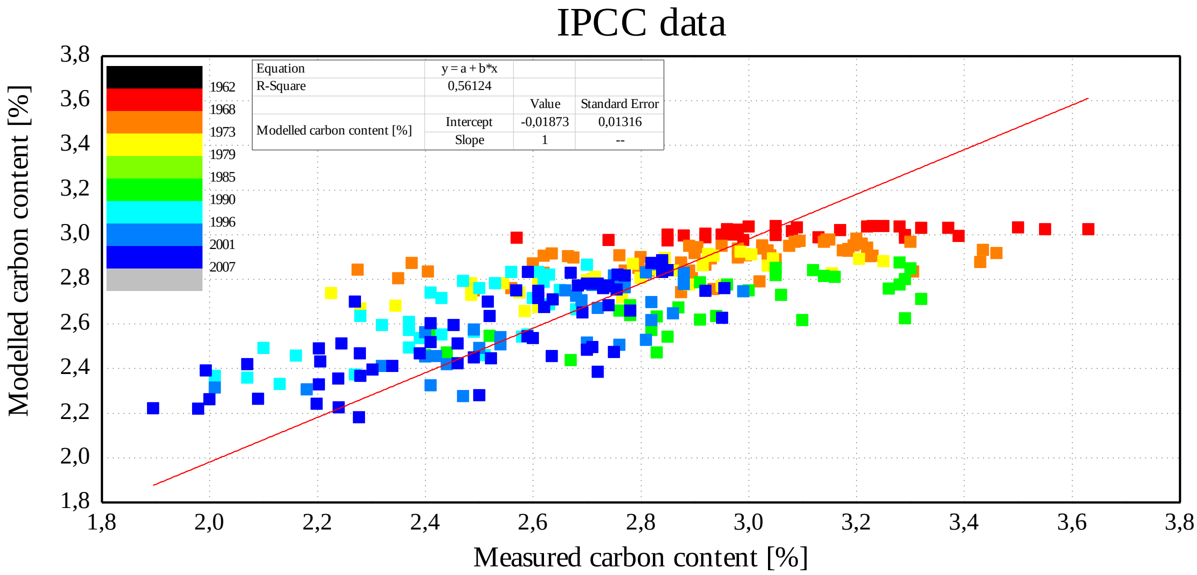


Figure 8.1. Crop-specific relations between harvested biomass (grains, seeds, tubers, roots, aboveground biomass) and the input from aboveground and belowground residues for both the IPCC and Nordic-limited dataset.

8.2 CHANGES IN SOIL CARBON CONTENT

Model prediction

Prediction power of the models used was tested by means of a coefficient of determination (R2). Figure 8.2 shows that the prediction power is reasonably high with 0.56 for both Nordic and IPCC data. The prediction based on Nordic data gives a somewhat better fit. Both calibrations underestimate the soil carbon content, but only by 0.019 %-units (IPCC) and 0,024 %-units (Nordic) on average.



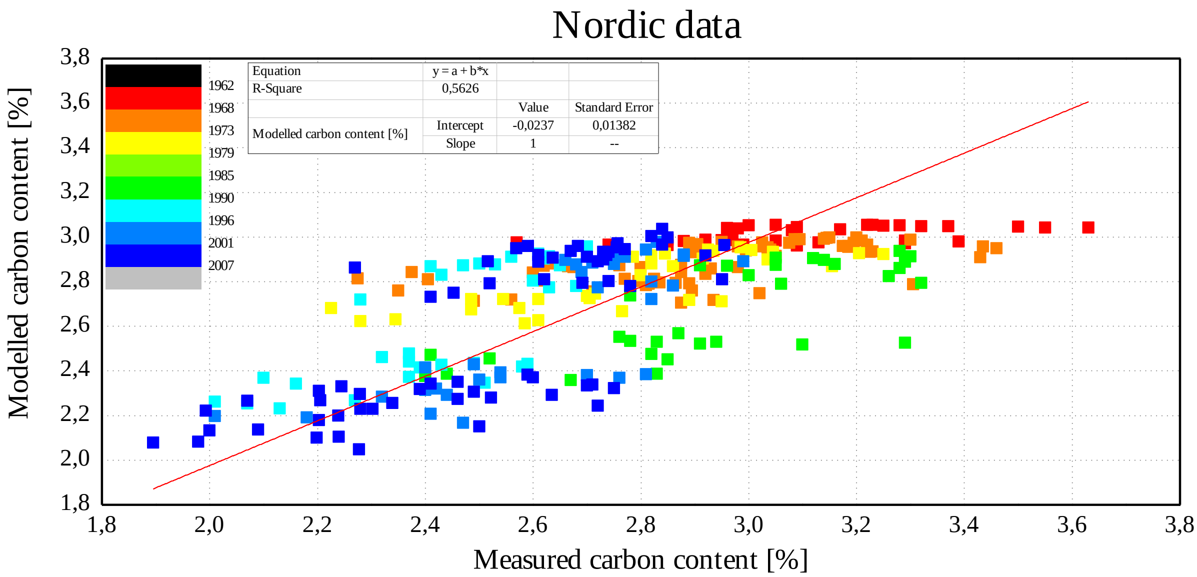


Figure 8.2. Comparison of the measured carbon content [%] of the long-term trial data set from Ekebo against a polynomic regression of the predicted carbon content [%] for IPPC data (top) and Nordic-limited data (bottom) used for crop residue calculation. The red line represents a linear fit calculated with a fixed slope (=1).

Annual soil carbon changes

Annual soil carbon changes were calculated as average carbon changes over a certain time interval. IPCC suggest a 20 year timespan. In the present study, however, a timespan of 40 years was used in the base case, as this was considered to better reflect the long time periods needed to achieve steady state levels in soil organic matter.

Annual soil carbon changes were calculated for each crop, assuming the same crop being cultivated continuously (Table 8.3). Though being an unrealistic assumption, results show the capacity for each crop to influence soil carbon on a comparable basis.

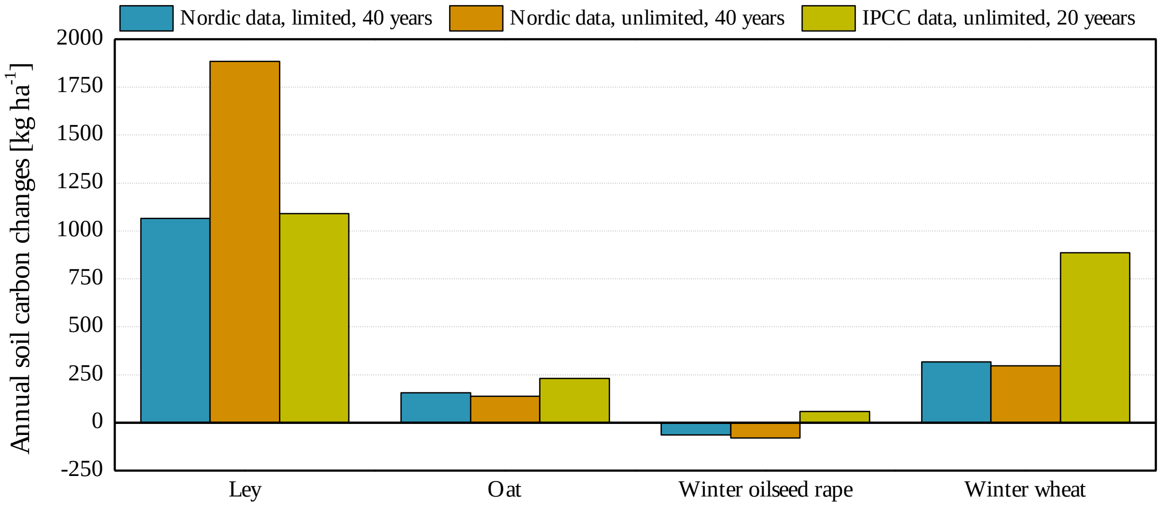


Figure 8.3. Annual soil carbon changes as a 40-year average according to the model calibretion for Nordic data (limited: blue columns; unlimited: orange columns) and as a 20-year average according to the model calibration on IPCC data (green columns) for crops investigated in this study. Continuous cultivation of each crop is assumed.

Though the effect of ley crops on soil carbon differs considerably between datasets (1 065-1 884 kg C ha^{-1}a^{-1}), it is clear that ley has a substantial positive effect on soil carbon content. In comparison, oat and winter oilseed rape have a negligible impact. Winter wheat has a positive effect on soil carbon content, but the effect from the IPCC dataset, where straw amounts are given as much higher, is three times as large as those from the Nordic datasets.

Annual soil carbon changes as calculated for each scenario are presented in Table 8.4. IPCC datasets clearly result in considerably higher soil carbon inputs than the Nordic datasets. This becomes clear in Scenario A and B where wheat stands for 50% of the cultivated area.

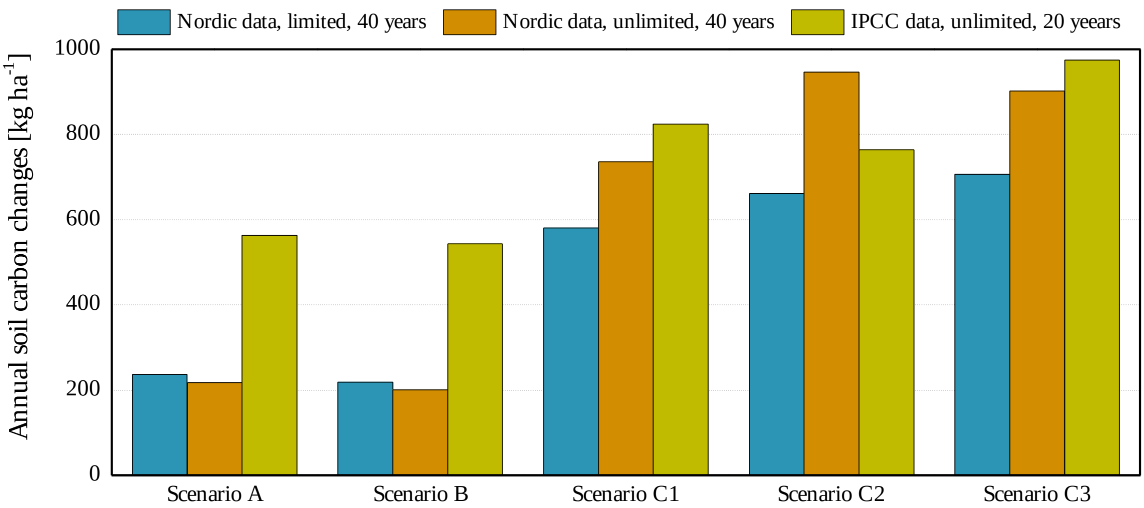


Figure 8.4. Annual soil carbon changes as a 40-year average according to the model calibre-tion for Nordic data (limited: blue columns; unlimited: orange columns) and as a 20-year average according to the model calibration on IPCC data (green columns) in the scenarios investigated in this study

Scenarios C1-3 result in similarly high soil carbon inputs for the different datasets. The substantial effect of ley for the Nordic datasets is set off against the high straw effect in the IPCC dataset. The additional soil car-bon input from a second year of ley (Scenario C2) is relatively low in comparison to Scenarios C1 and C3 with one year of ley in the crop rotation.

The sensitivity of annual carbon input due to biomass yield for crops and scenarios is shown in Figure 8.5 and Figure 8.6, respectively. This is relevant in calculations where the potential of yield in-creases. However, that step was not taken in the present study, and the potential of yield increases as an effect of soil carbon increases is only discussed qualitatively (Chapter 7.3). In Figure 8.6, it can be seen that the discussed yield increase of 20% would increase the annual carbon input from crop residues of about 20% for the C-scenarios.

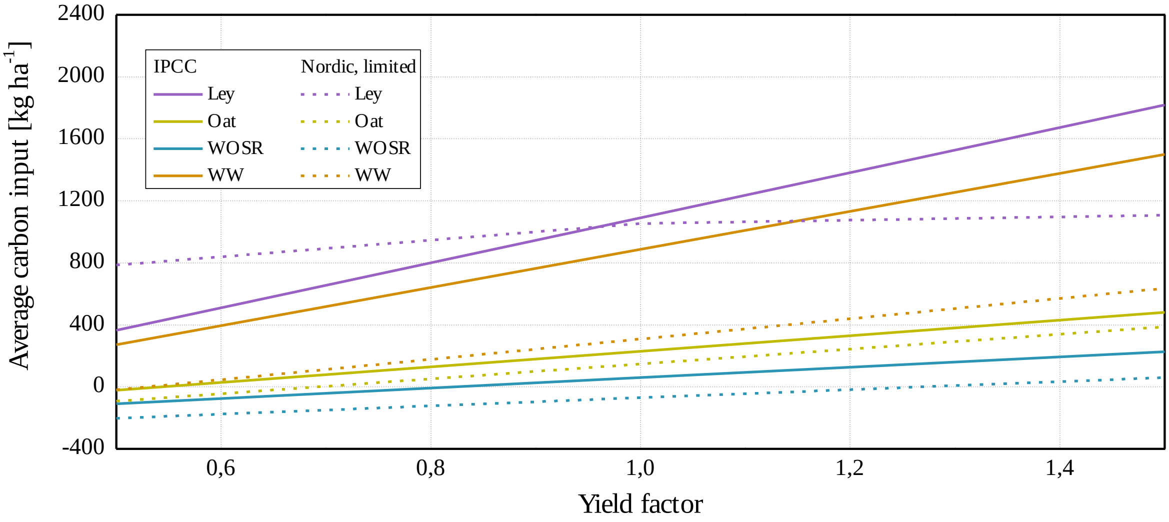


Figure 8.5. Sensitivity of soil carbon changes of crops due to changes in crop yield.

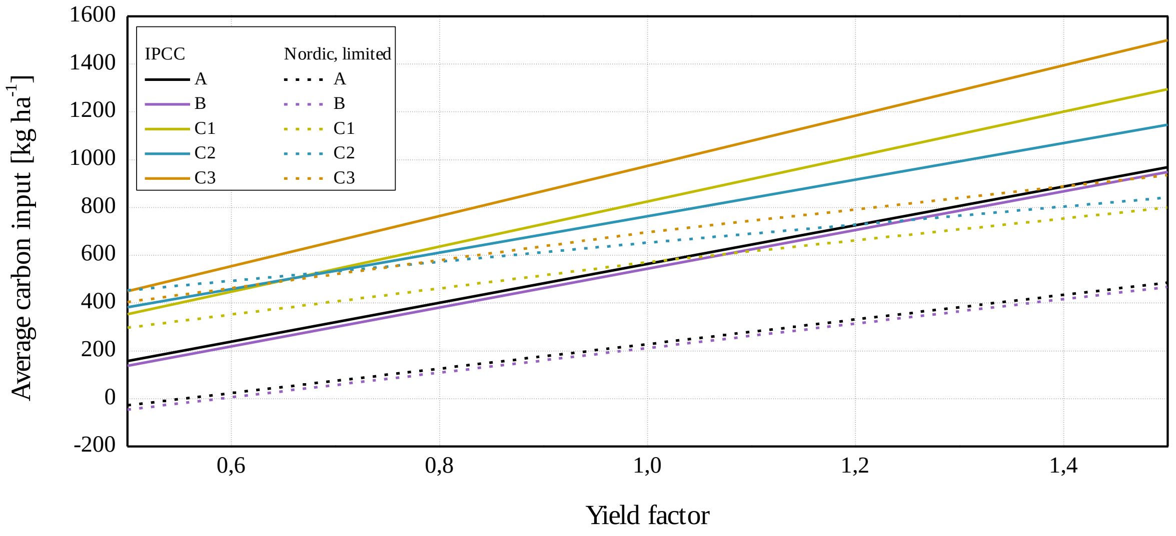


Figure 8.6. Sensitivity of soil carbon changes of scenarios due to changes in crop yield.

The three different data sets for amounts of crop residues and the two time intervals used for calculating average annual emissions all give different results on annual soil carbon changes. The results per scenario expressed as CO_2-eq/ha, yr. are summarized in Table 8.2.

Table 8.2 Soil carbon changes expressed as emissions of CO_2-eq/ha, yr. Base case (bold) and results of sensitivity analysis on input data for yield of crop residues.

		Scenario				
Calculation method crop residues	Average over (years)	A	B	C1	C2	C3
Base case – Nordic limited	**40**	**-869**	**-803**	**-2 128**	**-2 425**	**-2 592**
Nordic unlimited	40	-799	-734	-2 696	-3 472	-3 308
IPCC	40	-1 893	-1 828	-2 776	-2 624	-3 290
Nordic limited	20	-939	-865	-2 331	-2 649	-2 793
Nordic unlimited	20	-866	-793	-2 973	-3 866	-3 564
IPCC	20	-2 066	-1 993	-3 024	-2 803	-3 573

8.3 NITROUS OXIDE EMISSIONS AT FIELD APPLICATION

In the sensitivity analysis, the base case biogenic N_2O emission factor for emissions after soil application for manure, 2.5%, is replaced by the IPCC default emission factor of 1%. See Appendix C for details on calculations and references. Emission factors for digestate application are kept unchanged. In a second comparison, emission factors based on Swedish experimental data for pig manure (0.9%) and digestate application (0.2%) are used. The value for digestate is for digested cow manure. The effect of the change expressed as CO_2-eq/ha, yr. per scenario is shown inTable 8.3.

Table 8.3. Biogenic N_2O emissions at soil application expressed as emissions of CO_2-eq/ha, yr. Base case (bold) and results of sensitivity analysis on N_2O emission factors.

	Scenario				
Emission factor	A	B	C1	C2	C3
Base case – IPCC and National default	**373**	**155**	**409**	**563**	**489**
IPCC default	149	155	409	523	489
National experimental	134	31	82	113	98

8.4 METHANE EMISSIONS DURING STORAGE

In the base case, a methane conversion factor (MCF) of 3.5% is applied during storage of both pig manure and digestate. In a sensitivity analysis, Swedish experimentally derived data are applied, where an MCF of 2.8% during storage of pig manure, and 12% during digestate storage have been shown. The emission from digestate is from digestion of manly cow manure. This type of MCF-based calculation will affect the methane emissions based on the content of non-degraded organic material in the digestate. In addition, the methane leakage based on the total methane production in each scenario is calculated, where the methane leakage in the digestate storage is assumed to corresponds to 4% of the produced and upgraded methane. Recalculated as MCF, such an emission would correspond to 6.4% for digested pig manure and 18-20% for digestate from pig manure and ley. See Appendix D for details and references. The results of these two alternative calculations methods are shown in Table 8.4.

Table 8.4. Methane emissions during storage of manure or digestate expressed as emissions of CO_2-eq/ha, yr. Base case (bold) and results of sensitivity analysis.

	Scenario				
Calculation method	A	B	C1	C2	C3
Base case – MCF 3.5%	**145**	**56**	**87**	**130**	**98**
MCF 2.8% for manure and 12% for digestate	116	191	299	447	334
Leakage of 4% of produced methane from digestate	145	102	448	723	560

8.5 ALTERNATIVE PROCESS ENERGY IN BIOGAS PRODUCTION

To use natural gas for heating of the biogas process is not the most climate efficient method, and is quite unconventional in Swedish biogas plants. Thus, replacing natural gas for heating of the biogas process with solid biofuels is a realistic scenario. The emissions expressed as CO_2-eq/ha, yr. per scenario from only natural gas use in the base case are shown in Table 8.5 together with the impact of instead using solid biofuels.

Table 8.5. Impact of replacing natural gas for heat production in the biogas process (base case, bold) with solid biofuels.

Fuel for heating	B	C1	C2	C3
Base case (natural gas)	**71**	**120**	**159**	**138**
Solid biofuels	3	5	6	5

The electricity used in the biogas process (extrusion of ley, pumping, stirring, upgrading and compression) is in the base case assumed to be a Swedish average electricity mix. In the EU renewable energy directive, instead, a Nordic mix with a higher climate impact is to be used for calculation of emissions. Thus, the effect on this system of replacing the average life cycle emissions for Swedish electricity mix by Nordic mix is calculated expressed as CO_2-eq/ha, yr. per scenario (Table 8.6).

Table 8.6. Impact of replacing Swedish electricity mix in biogas production and upgrading (base case, bold) with Nordic electricity mix.

Origin of electricity	B	C1	C2	C3
Base case (Swedish mix)	**8**	**24**	**36**	**29**
Nordic mix	26	81	125	100

8.6 SUMMARY AND DISCUSSION

The net result on GHG emissions for each of the introduced changes (Table 7.2) is shown as the base case in the following figures together with the effect changing input data as described in the previous chapters. The effects are shown as net avoided GHG emissions (negative values) of the above described sensitivity analyses and are summarized in Figure 8.7-Figure 8.10.

The effects of alternative data when calculating crop residues are shown in Figure 8.7. The amounts of crop residues will impact both N_2O emissions from crop residues and soil carbon build-up. These two impacts are in Figure 8.7 first shown separately and then as the total impact. For the C scenarios, higher relative amounts of crop residues (Nordic unlimited) will give increased N_2O emissions, reducing the climate benefit, but higher soil carbon build-up, improving the climate benefit. The total result of increased amounts of crop residues (Nordic unlimited) is an improved climate benefit. The IPCC data, however, gives a relative decrease in amounts of crop residues when ley is introduced, reducing the climate benefit.

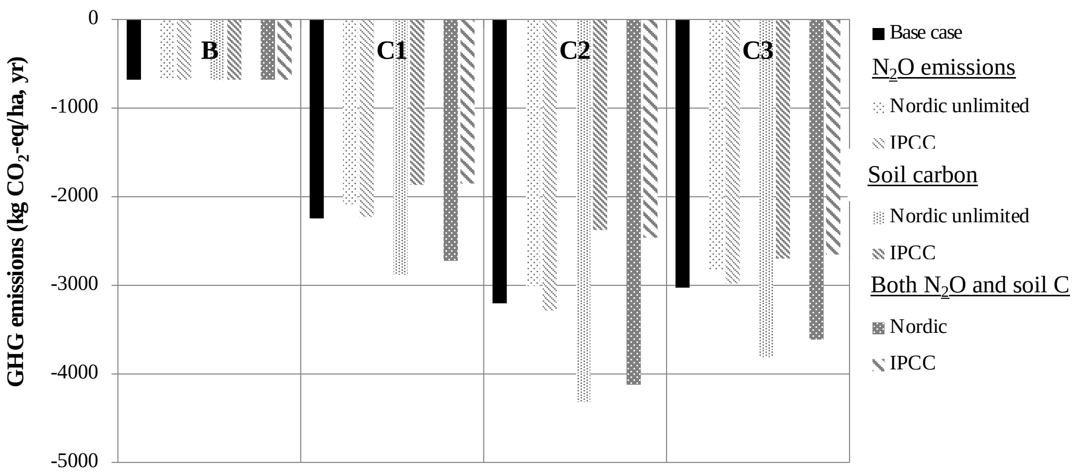


Figure 8.7. Result of sensitivity analyses on calculation method for amounts of crop residues.

The impact on annual soil carbon build-up of the time perspective of 40 years (as in the base case) and 20 years is shown in Figure 8.8. The time perspective evaluation is also done for the three data sets on amounts of crop residues. To use the shorter time period of 20 years to calculate the annual soil carbon build up will give a higher annual contribution since the build-up is as highest in the first years, so will improve the GHG benefit especially for the C-scenarios. However, since reaching a steady state was shown to be a slow process, using a longer time perspective is more relevant.

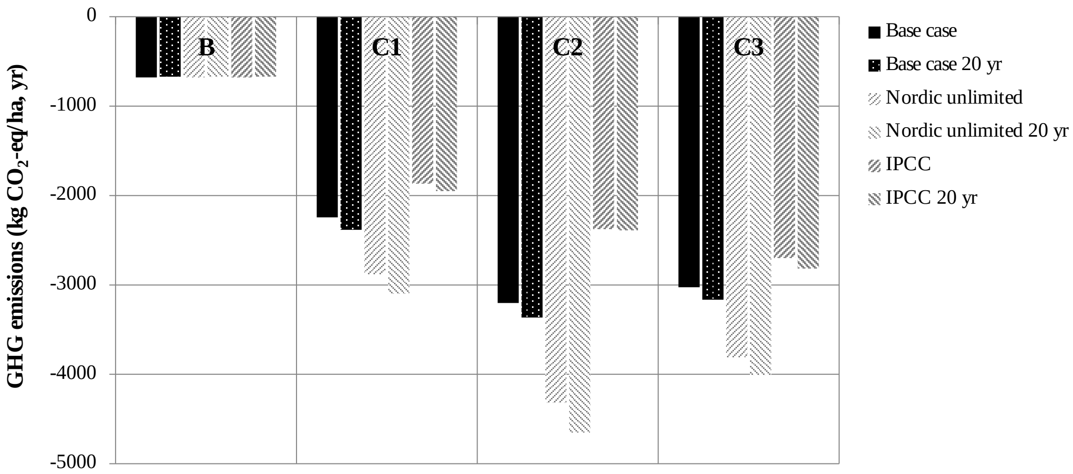


Figure 8.8. Result of sensitivity analyses on time perspective for soil carbon model.

The impact of varying GHG emissions from field application of biofertilizer or methane emissions from storage in relation to the reference systems is shown in Figure 8.9.

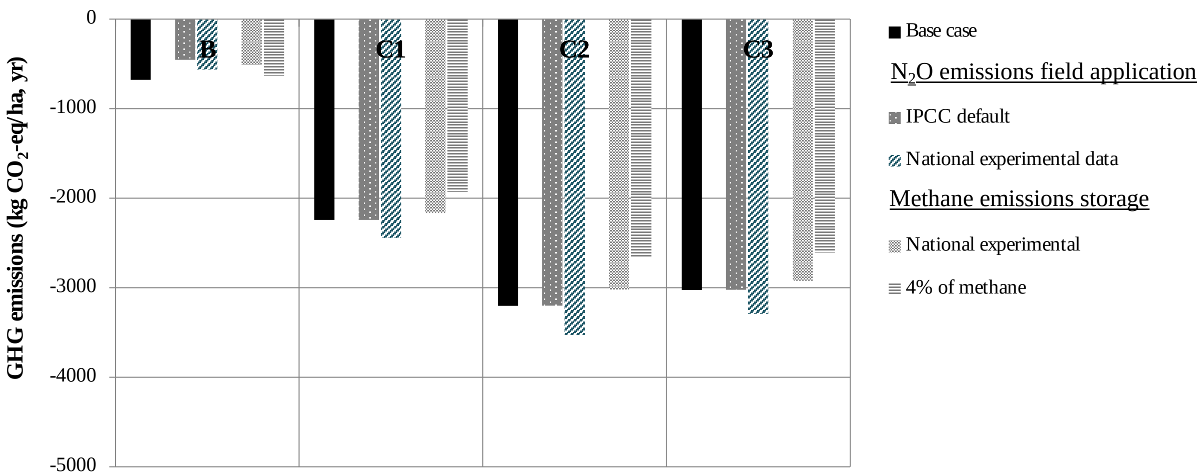


Figure 8.9. Result of sensitivity analyses on biogenic emissions of N_2O and methane from storage and field application.

Finally, the impact of changing input data related to the energy supply in the biogas production is summarized in Figure 8.10.

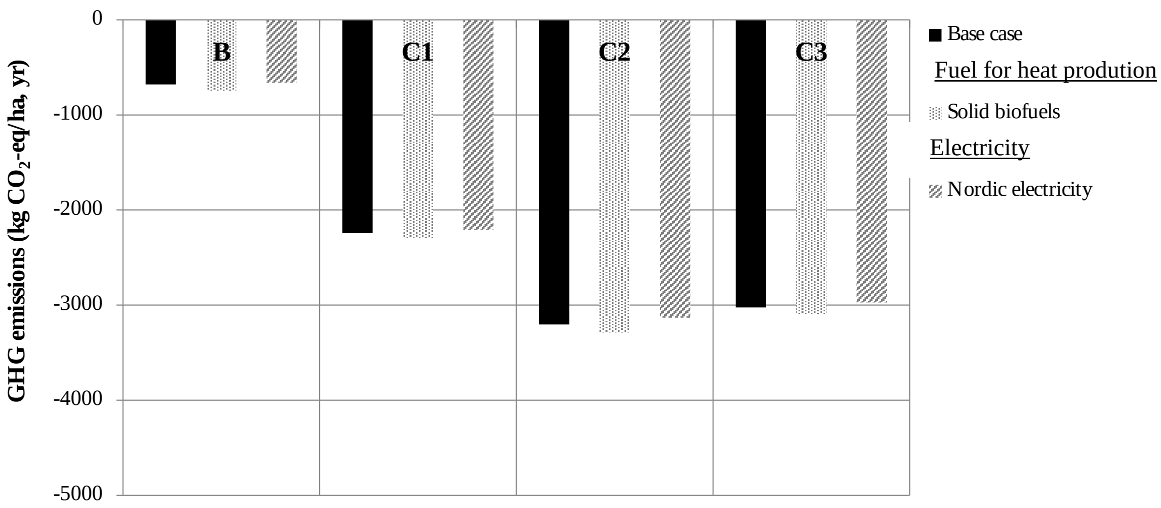


Figure 8.10. Result of sensitivity analyses on process energy in biogas plant.

When biogas is produced only from manure, as in scenario B, the sensitivity analysis shows that changes in emissions of N_2O compared to emissions in conventional manure handling, potentially has a large negative impact on the result (Figure 8.9). The net GHG emissions could then decrease to -456 kg CO_2-eq/ha, yr. The biogenic N_2O emission from biofertilizer field application is a value with large uncertainty. However, to use the same emission factor for both manure and digestate, as in this sensitivity analysis, does not agree with the theory behind the microbial processes leading to this emission. The removal of organic carbon from the biofertilizer is likely to have a positive impact on reducing N_2O emissions after field application. This has also been experimentally shown in Swedish trials, reflected in a use of national experimental data (Figure 8.9).

For the production of biogas from manure (scenario B), the sensitivity analysis shows that replacing the fuel for heating of the biogas process from natural gas to solid biofuels has the largest positive impact on emissions, increasing avoided GHG emissions to -747 kg CO_2-eq/ha, yr. (Figure 8.10). Manure is a bulky biogas feedstock, where large volumes are to be heated. Thus, process improvements based on better GHG efficiency per weight of feedstock are important.

For the scenarios where ley is used for biogas production in addition to manure (Scenarios C1-C3), the sensitivity analysis shows that a change to using of the IPPC calculation method for amounts of crop residues would result in the lowest GHG efficiency (Figure 8.7). The net avoided GHG emissions decrease to -1 852, -2 390 and -2 653 CO_2-eq/ha, yr. respectively for scenarios C1, C2 and C3. The IPCC method gives very high straw amounts in relation to the Nordic data, giving increased amounts of crop residues in the A and B scenario when share of cereal cultivation is high. This is shown as contribution to annual soil carbon changes in Figure 8.4. These high straw yields presented by IPCC are, however, not realistic to apply to present cereal cultivation in Sweden.

The highest impact on GHG efficiency for the ley based systems (The C-scenarios) is shown in the sensitivity analysis when the restriction on the amount of root biomass is removed and the time perspective in soil carbon calculation is changed from 40 to 20 years. The net avoided GHG emissions increase to -3 098, -4 653 and -4 007 kg CO_2-eq/ha, yr. respectively for scenarios C1, C2 and C3. However, at the high ley yields on good soils in the present study, it is not seen as realistic to achieve so high root biomass amounts, which is why the limitation was introduced. The chosen data on soil carbon contribution from crop residues give net emissions that are in between the above described extremes, and is considered a realistic alternative.

The base case and the highest and lowest results based on sensitivity assessments are summarized in Table 8.7. The result of the extreme values from the sensitivity analysis expressed also as GHG emission per GJ fuel is shown in Figure 8.11.

Table 8.7. Net GHG emissions per hectare and year. Base case emission and maximum and minimum values.

Scenario	Emission		
	kg CO_2-eq/ha, yr		
	Base case	max	Min
B	-679	-747	-456
C1	-2 244	-3 098	-1 852
C2	-3 203	-4 653	-2 390
C3	-3 025	-4 007	-2 653

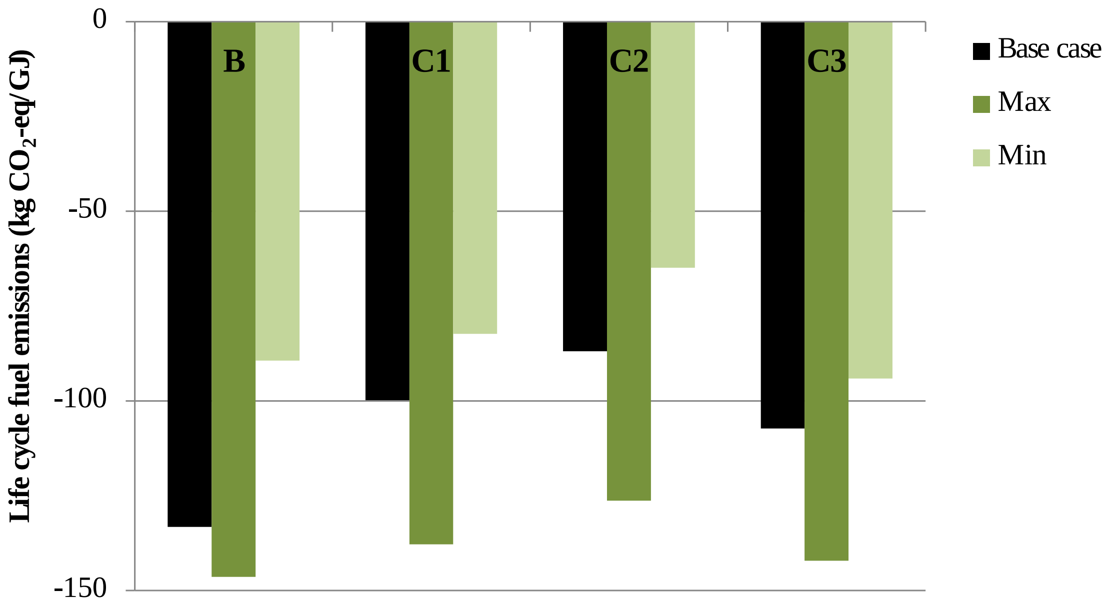


Figure 8.11. Net result from Table 8.7 calculated as life cycle emission per GJ fuel including end use.

9 CONCLUSIONS

On Wrams Gunnarstorp, the model farm in the farm based case study, soil compaction on the medium to heavy clay soils is a problem. The crop yields, based on annual measurements on the farm, are 5% (winter wheat), 15% (oats) and 20% (winter rapeseed) lower than average yields for the region. Aware of the problem, three years of meadow fescue for seed production has been integrated in the cereal based crop rotation on the most problematic soils. However, the market for grass seeds being limited, the economic possibilities for integrating ley crops in other parts of the crop rotation is limited in a region with little demand for cattle feed. The approach in the present study was to integrate 1-2 years of ley crops in the crop rotation and to use this as feedstock for biogas production. The effects of this on GHG emissions, soil organic matter and food crop production were evaluated.

The climate benefit is high for all the investigated scenarios where ley is introduced in the crop rotation and used for biogas production. Introducing a change where 20-33% of the 650 ha farm is used for production of ley as a biogas crop and the produced biogas is used to replace fossil vehicle fuels will result in avoided GHG emission of 1 460-2 080 ton CO_2-eq/year for the 650 ha farm. If instead the GHG emissions are presented per GJ of biogas, the emission reduction amounts to 106% to 128% when the produced upgraded and compressed biogas replaces fossil fuels for transport.

The climate benefit for scenarios with ley production is to a large extent the effects from replacing fossil fuels with the biogas produced; 480-870 l petrol/ha, yr. or, on average over the whole 650 ha farm, 15-24 TJ/yr. Equally important is the effect on increased soil organic matter content on farm level. Apart from the role as carbon sink and the impact on GHG emissions, the increase in soil carbon levels is important for long term soil fertility and productivity on this type of compacted clay soils. In the ley scenarios, the soil carbon content increases steadily from 2% today to 3% within 20-30 years to reach a steady state level of around 5%. Here, the possibility of using ley for biogas production opens up for a possibility of integrating ley in the crop rotation in cereal intensive areas even if there is no demand for cattle feed. Still, land would be taken out of food production. The impact of increased soil organic matter on soil compaction and fertility and the potential of increasing yields could compensate for a large part of this. In this study, the decrease in food crop production is compensated for by adding GHG emissions from the additional grain and oil seed cultivation outside the farm to fulfil an unchanged total output of food products. This is decreasing the climate benefit of the systems slightly. Despite this, the resulting reduction in GHG emission is large.

One important aspect in all ley scenarios is that the ley is undersown the year before the main harvest (called year 0), making ley biomass harvest possible in autumn year 0. This gives good land use efficiency, using the benefit of harvesting that extra biomass. The economic feasibility of this small harvest remains to be evaluated.

The sensitivity analysis shows that the ley scenarios are sensitive to the chosen data for calculation of amounts of crop residues. However, the IPCC data set, which gives very high straw amounts for cereals, is not valid for the actual conditions in the Nordic countries, and the calculation method developed within the project is considered to give a better estimation of actual conditions. The ley scenarios are also sensitive to calculations assuming high methane leakage from the digestate storage. The emissions evaluated in the sensitivity assessment are however high considering that the digestate is used as fertilizer in the period 1 April-15 May, and the share of the annual digestate production that is stored during the warm part of the year, May-October, is only 29% of the annual production. An aspect like this, the time for spreading, is important to consider when a digestate containing high amounts of organic matter, like in this case from ley crops with relatively low biodegradability, is produced.

The soil carbon contribution is important to consider in all systems where biomass is removed from farm land. The first step in the farm based case study was to assess the effect of using manure for biogas production, and then recycling the residue, the digestate, to the farm as biofertilizer. The climate benefit for this manure based biogas production is very good, but it has been argued that the impact on soil organic matter could be negative. However, the negative effect on soil carbon was in this study shown to be negligible. The main positive impact of biogas production from manure is when the biofuels replaces fossil transportation fuels, but the impact on reduced biogenic emissions of N_2O and methane is also important.

In the farm based case study performed in this project, it has been shown which features of the investigated scenarios that are most important for good GHG efficiency. At the same time, effects on soil organic matter content and food crop yields have been presented. The outcome is important in fulfilling the future criteria in sustainability certification of biofuel systems, such as avoiding indirect land use changes and maximizing GHG performance (Ahlgren & Börjesson, 2011). Both outcomes are equally important for improved understanding of soil fertility challenges and land use competition between food and energy crop production. Neither biogas production from manure nor from ley crops shows good profitability from a biogas plant perspective at present biofuels prices (Lantz et al., 2013), which might hinder the introduction of such systems in spite of the positive impact on GHG emissions. The intention of the researchers behind the present study is to follow up with a study encompassing several Swedish regions and including economic evaluations of ley as biogas feedstock, including aspects that are important from the farm perspective as effects of preceding crop and soil carbon contribution.

REFERENCES

Ahlgren, S., Börjesson, P. 2011. Indirekt förändrad markanvändning och biodrivmedel – en kunskapsöversikt. Rapport No. 73, Environmental and Energy Systems Studies, Department of Technology and Society, Lund University, Sweden.

Ahlgren, S., Hansson, P.-A., Kimming, M., Aronsson, P., Lundkvist, H. 2011. Greenhouse gas emissions from cultivation of agricultural crops for biofuels and production of biogas from manure. Implementation of the directive of the European parliament and of the council on the promotion of the use of energy from renewable sources. Dnr SLU ua 12-4067/08, revised 8 Sep 2009, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Akhtar, M., Mashkoor Alam, M. 1992. Effect of crop residues amendments to soil for the control of plant-parasitic nematodes. *Bioresource Technology*, **41**(1), 81-83.

Anderson-Teixeira, K., Davis, S., Masters, M., DeLucia, E. 2009. Changes in soil organic carbon under biofuel crops. *Global Change Biology Bioenergy*, **1**, 75-96.

Andrén, O., Kätterer, T. 1997. ICBM: The Introductory Carbon Balance Model for Exploration of Soil Carbon Balances. *Ecological Applications*, **7**(4), 1226-1236.

Andrén, O., Kätterer, T., Karlsson, T., Eriksson, J. 2008. Soil C balances in Swedish agricultural soils 1990–2004, with preliminary projections. *Nutr Cycl Agroecosyst*, **81**, 129-144.

Arp, B., Hänsel, M., Karalus, W., Kolbe, H., Schuster, M., Jäckel, U. 2010. Ölfrüchte im Ökologischen Landbau. Sächsisches Landesamt für Umwelt, Landwirtschaft und Geologie.

Arvidsson J., 2008. Soil compaction related to spreading manure – A calculation model for the manure programme STANK. Department of Soil and Environmental Science, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Arvidsson, J., Håkansson, I. 1991. A model for estimating crop yield losses caused by soil compaction. *Soil Tillage Research*, **20**, 319-332.

Becka, D., Vasák, J., Kroutil, P., Stranc, P. 2004. Autumn growth and development of different winter oilseed rape variety types at three input levels. *Plant Soil Environment*, **50**(4), 168-174.

Bertilsson G. 2006. Perspectives on soil carbon. http://www.greengard.se/cpersp.htm. Visited 2013-08-27.

Bertilsson G. 2009. Odlingsperspektiv, bakgrund och dokumentation. http://www.greengard.se/Odlingsperspdok090716.htm. Visited 2013-08-27.

Bioenergiportalen. 2013. www.bioenergiportalen.se. Visited 2013-03-21.

Björnsson, L. 2013. Energigrödor för biogasproduktion. Del 3, energi och växthusgaseffektivtet. Rapport No. 82, Environmental and Energy Systems Studies, Department of Technology and Society, Lund University, Sweden.

Bolinder, M.A., Janzen, H.H., Gregorich, E.G., Angers, D.A., VandenBygaart, A.J. 2007. An approach for estimating net primary productivity and annual carbon inputs to soil for common agricultural crops in Canada. *Agriculture, Ecosystems & Environment*, **118**(1–4), 29-42.

Börjesson, P., Tufvesson, L., Lantz, M. 2010. Livscykelanalys av svenska biodrivmedel. Rapport No. 70, Environmental and Energy Systems Studies, Department of Technology and Society, Lund University, Sweden.

Börjesson, P., Prade, T., Lantz, M., Björnsson, L. 2013. Energy crop-based biogas as vehicle fuel – differences in energy efficiency and greenhouse gas performance. *Manuscript submitted for publication in Biomass & Bioenergy*.

Carlsson, M., Uldal, M. 2009. Substrathandbok för biogasproduktion. SGC Rapport 200. Svenskt Gastekniskt Ceter, Malmö, Sweden.

Clemens, J., Trimborn, M., Weiland, P., Amon, B. 2006. Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. *Agriculture, Ecosystems and Environment*, **112**, 171-177.

EC. 2002. Regulation (EC) No 1774/2002 of the European parliament and of the council of 3 October 2002 laying down health rules concerning animal by-products not intended for human consumption, Official Journal of the European Communities.

Energimyndigheten. 2011. Vägledning till regelverket om hållbarhetskriterier för biodrivmedel och flytande biobränslen, Report 2011:23, Swedish Energy Agency, Eskilstuna, Sweden.

EU. 2009. Directive 2009/28/EC of the European parliament and of the council on the promotion of the use of energy from renewable sources.

FNR. 2010. Biogas Messprogramm II - 61 Biogasanlagen im Vergleich. Bundesministerium fur Ernährung, Landwirtshafts und Verbraucherschutz, Berlin, Germany.

Foged, H. L., Hansen, M. M., Möller, H. B. 2012. Extrusion of solid plant biomass. Technology contributing to a sustainable and profitable biogas production. Report, Project Baltic Compass. Agro Business Park A/S, Tjele, Denmark.

Forster, P., V. Ramaswamy, P. Artaxo, T. Berntsen, R. Betts, D.W. Fahey, J. Haywood, J. Lean, D.C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schulz and Van Dorland, R. 2007. Changes in Atmospheric Constituents and in Radiative Forcing. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Giles, M., Morley, N., Baggs, E.M., Daniell, T.J. 2012. Soil nitrate reducing processes - drivers, mechanisms for spatial variation, and significance for nitrous oxide production. in: *Frontiers in Microbiology*, Vol. 18, pp. 407.

Gissén, C., Prade, T., Kreuger, E., Nges, I.A., Rosenqvist, H., Svensson, S.-E., Lantz, M., Mattsson, J.E., Börjesson, P., Björnsson, L. 2014. Comparing energy crops for biogas production –

yields, energy input and costs in cultivation using digestate and mineral fertilisation. *In press, Biomass & Bioenergy*.

Gode, J., Martinsson, F., Hagberg, L., Öman, A., Höglund, J. and Palm, D., 2011. Miljöfaktaboken 2011-Uppskattade emissionsfaktorer för bränslen, el, värme och transporter, Thermal Engineering Research Institute, Stockholm, Sweden.

GreppaNäringen. 2011. Stallgödselkalkyl version 2011-03-31. http://www.greppa.nu/stallgodsel/varderadinstallgodsel.4.1c0ae76117773233f780001828.html. Visited 2012-11-07.

van den Heuvel, R.N., Bakker, S.E., Jetten, M.S., Hefting, M.M. 2011. Decreased N2O reduction by low soil pH causes high N2Oemissions in a riparian ecosystem. *Geobiology*, **9**(3), 294-300.

Huang, G.F., Wu, Q.T., Wong, J.W.C., Nagar, B.B. 2006. Transformation of organic matter during co-composting of pig manure with sawdust. *Bioresource Technology*, 1834-1842.

Hutchings, N.J., Sommer, S.G., Anderssen, J.M., Asman, W.A.H. 2001. A detailed ammonia emission inventory for Denmark. *Asmospheric Environment*, **35**, 1959-1968.

IEA. 2012. CO_2 emissions from fuel combustion 2012 edition, International Energy Agency, Paris, France.

IPCC, The Intergovernmental Panel on Climate Change. 2006. IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Institute for Global Environmental Strategies, Japan.

ISO, 2006. Environmental management – Life cycle assessment – Requirements and guidelines, ISO 624 14044.

Johnsson, H., Larsson, M., Lindsjö, A., Mårtensson, K., Persson, K., Torstensson, G. 2008. Läckage av näringsämnen från svensk åkermark, Naturvårdsverket, Stockholm, Sweden.

Jordbruksverket. 2007. Gödsel och miljö. Swedish Board of Agriculture, Jönköping, Sweden.

JRC, 2011. WELL-to-Wheels Analysis of future automotive fuels and power trains in the European context, Report 3c, July 2011, Institute for Environment, European Commission Joint Research Center.

Karlsson, S., Rodhe, L. 2002. Översyn av Statistiska Centralbyråns beräkning av ammoniakavgången i jordbruket – emissionsfaktorer för ammoniak vid lagring och spridning av stallgödsel. JTI – Institutet för jordbruks- och miljöteknik, Uppsala, Sweden.

Kasimir Klemedtsson, Å. 2001. Metodik för skattning av jordbrukets emissioner av lustgas: underlag för Sveriges nationalrapport till Klimatkonventionen. Swedish Environmental Protection Agency, Stockholm. Sweden.

Kirchmann, H., Eriksson, J., Snäll, S. 1999. Properties and Classification of Soils of the Swedish Long-Term Fertility Experiments: IV. Sites at Ekebo and Fjärdingslöv. *Acta Agriculturae Scandinavica, Section B - Soil & Plant Science*, **49**(1), 25-38.

Koga, N., Smith, P., Yeluripati, J.B., Shirato, Y., Kimura, S.D., Nemoto, M. 2011. Estimating net primary production and annual plant carbon inputs, and modelling future changes in soil carbon stocks in arable farmlands of northern Japan. *Agriculture, Ecosystems & Environment*, **144**(1), 51-60.

KTBL. Kuratorium fur Technik und Bauwesen in der Landwirtschaft. 2013. Wirtschaftlichkeitsrechner Biogas. http://daten.ktbl.de/biogas/startseite.do?zustandReq=1&selectedAction=showSubstrate#anwendung Visited 2013-03-21.

Kätterer, T., Andrén, O. 2001. The ICBM family of analytically solved models of soil carbon, nitrogen and microbial biomass dynamics — descriptions and application examples. *Ecological Modelling*, **136**(2–3), 191-207.

Kätterer, T., Bolinder, M.A., Andrén, O., Kirchmann, H., Menichetti, L. 2011. Roots contribute more to refractory soil organic matter than above-ground crop residues, as revealed by a long-term field experiment. *Agriculture, Ecosystems & Environment*, **141**(1–2), 184-192.

Lantz, M., Kreuger, E., Björnsson, L. 2013. Impact of energy crop selection on process parameters and economy in the production of biogas as vehicle fuel. *Manuscript submitted for publication in Biomass and Bioenergy.* Available as manuscript in the PhD thesis of Lantz (2013).

Lantz, M. 2013. Biogas in Sweden - Opportunities and challenges from a systems perspective. Doktorsavhandling. Miljö- och Energisystem, Lunds Tekniska Högskola, Lunds Universitet. ISBN 978-91-7473-469-0.

Lantz, M., Börjesson, P. 2013. Greenhouse gas and energy assessment of the biogas from co-digestion injected into the natural gas grid. A Swedish case-study including effects on soil properties. *Manuscript submitted for publication in Renewable Energy.* Available as manuscript in the PhD thesis of Lantz (2013).

Lantz, M., Ekman, A., Börjesson, P. 2009. Systemoptimerad produktion av fordonsgas. Rapport No. 69, Environmental and Energy Systems Studies, Department of Technology and Society, Lund University, Sweden.

Lehmann. 2013. Information supplied by supplier of equipment. http://www.lehmann-maschinenbau.de/en/biogas-technology/bio-extrusion.html. Lehmann Maschinenbau GmbH. Visited 2013-05-13.

Liebetrau, J., Reinelt, T., Clemens, J., Hafermann, C., Friehe, J., Weiland, P. 2013. Analysis of greenhouse gas emissions from 10 biogas plants within the agricultural sector. *Water Science & Technology*, **67**(6), 1370-1379.

Maag, M., Vinther, F.P. 1996. Nitrous oxide emissions by nitrification and denitrification in different soil types and at different soil moisture contents and temperatures. *Applied Soil Ecology*, **4**, 5-14.

Mangino, J., Bartram, D., Brazy, A. 2002. Development of a methane conversion factor to estimate emissions from animal waste lagoons. in: *US EPA 17th Annual Emission Inventory Conference*. Atlanta GA, US.

McCarty, P. 1964. Anaerobic waste treatment fundamentals, Part one, Chemistry and microbiology. *Public Works*, **95**, 107-112.

Möller, K., Stinner, W. 2009. Effects of different manuring systems with and without biogas digestion on soil mineral nitrogen content and on gaseous nitrogen losses (ammonia, nitrous oxides). *European Journal of Agronomy*, **30**(1), 1-16.

Naturvårdsverket. 2013. National inventory report Sweden 2013. Naturvårdsverket, Stockholm, Sweden.

Nges, I., Escobar, F., Fu, X., Björnsson, L. 2011. Benefits of supplementing an industrial waste anaerobic digester with energy crops for increased biogas production. . *Waste Management*, **32**, 53-59.

Nilsson, D., Bernesson, S. 2009. Halm som bränsle : Del 1: Tillgångar och skördetidpunkter. Department of Energy and Technology, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Persson, T., Bergkvist, G., Kätterer, T. 2008. Long-term effects of crop rotations with and without perennial leys on soil carbon stocks and grain yields of winter wheat. *Nutr Cycl Agroecosyst*, **81**, 193-202.

Prade, T., Svensson, S.-E., Mattsson, J.E. 2012. Energy balances for biogas and solid biofuel production from industrial hemp. *Biomass & Bioenergy*, **40**, 36-52.

Pietola, L., Alakukku, L. 2005. Root growth dynamics and biomass input by Nordic annual field crops. *Agriculture, Ecosystems & Environment*, **108**(2), 135-144.

Rasmusen, H. 2008. Wrams Gunnarstorp. Personal communcation.

Rasmusen, H. 2013. Wrams Gunnarstorp. Personal communcation.

Rodhe, L., Ascue Contreras, J., Tersmeden, M., Ringmar, A., Nordberg, Å. 2008. Växthusgasemissioner från lager med nötflytgödsel. Rapport R370. Institutet för jordbruks- och miljöteknik, Uppsala, Sweden.

Rodhe, L., Abubaker, J., Ascue, J., Pell, M., Nordberg, Å. 2012. Greenhouse gas emissions from pig slurry during storage and after field application in northern European conditions. *Biosystems Engineering*, **113**, 379-394.

Rodhe, L., Ascue, J., Tersmeden, M., Willén, A., Nordberg, Å, Salomon, E., Sundberg, M. 2013. Växthusgaser från rötad och orötad nötflytgödsel vid lagring och efter spridning. JTI-rapport 2013, Lantbruk & industri Nr 413. Institutet för jordbruks- och miljöteknik, Uppsala, Sweden.

Röing, K., Andrén, O., Mattsson, L. 2005. Long-term management effects on plant N uptake and topsoil carbon levels in Swedish long-term field experiments: cereals and ley, crop residue treatment and fertiliser N application. *Acta Agriculturae Scandinavia Section B – Soil and Plant*, **55**, 16-22.

SJV. 2011. Riktlinjer för gödsling och kalkning. Swedish Board of Agriculture, Jönköping, Sweden.

Soilservice. 2012. Soilservice - Conflicting demands of land use, soil biodiversity and the sustainable delivery of ecosystem goods and services in Europe. Final report, the FP7 project SOIL SERVICE.

Sommer, S.G., Petersen, S.O., Sogaard, H.T. 2000. Greenhouse gas emissions from stored livestock slurry. *Journal of Environmental quality*, **29**, 744-751.

Swedgas, 2012. Gas quality in Sweden, http://www.swedegas.se/vara_tjanster/gaskvalitet/gaskvalitet_i_sverige, Visited 2012-11-25.

Takaya, N., Catalan-Sakairi, M.A.B., Sakaguchi, Y., Kato, I., Zhou, Z., Shoun, H. 2003. Aerobic denitrifying bacteria that produce low levels of nitrous oxide. *Appl Environ Microbiol*, **69**(6), 3152-3157.

Tufvesson, L., Lantz, M., Björnsson, L. 2013. Miljönytta och samhällsekonomiskt värde vid produktion av biogas från gödsel. Rapport No. 86, Environmental and Energy Systems Studies, Department of Technology and Society, Lund University, Sweden.

UoT. 2008. Limestone Quarrying and Processing: A Life-Cycle Inventory. University of Tennessee, Center for Clean Products, USA.

Weier, K.L., Doran, J.W., Power, J.F., Walters, D.T. 1993. Denitrification and the dinitrogen/nitrous oxide tayio as affected by soil water, avaliable carbon and nitrate. *Soil Sci. Soc. Am. J.*, **57**, 66-72.

Wulf, S., Maeting, M., Clemens, J. 2002. Application technique and slurry co-fermentation effects on ammonia, nitrousn oxide and methane emissions after spreading: I. ammonia volatilization. *J Environmental Quality*, **31**, 1789-1794

APPENDIX A. BIOGAS PROCESS CALCULATIONS

For the present project, a calculation model has been applied for calculation of the changes that occur for a certain feedstock during the anaerobic digestion process. The input values for the different feedstock types will be the concentration of total nitrogen (N-tot), ammonia-nitrogen (NH_4-N), dry matter (DM), volatile solids (VS), carbon (C) content and the maximum methane yield (B_0) of the feedstock. Based on these inputs, the digestate features after anaerobic digestion at a certain methane yield can be calculated based on the model presented by Lantz et al (2013). The most important features of the calculations from that study are repeated here;

- DM, VS and C concentration in the digestate is calculated on the basis of the input feedstock values and the mass loss in form of biogas (C, H and O). The potential loss of compounds other than CH_4 and CO_2 through the biogas, e.g. water, hydrogen sulphide or ammonia, is not included in the calculations.

- The biogas production (released biogas) is calculated based on the CH_4 yields and assuming a biogas content of 55% CH_4 in all cases. The biogas produced is assumed to represent 95% of the metabolised mass of C, H and O with another 5% of the metabolised mass being assimilated into new microbial biomass (McCarty, 1964).

- The fraction of organic nitrogen converted to NH_4-N is calculated by assuming that the degree of nitrogen mineralization is equal to the degree of VS-metabolization. Since microbial biomass will also assimilate part of the mineralized N, reconverting it to organic N, the amount of NH_4-N is reduced by the corresponding amount.

A.1 CHANGES RELATED TO ORGANIC MATERIAL

A.1.1 Methane yield

The methane yield in the biogas process has impact on many of the other calculated outputs. For swine manure, no experimental studies were performed within the present study. Instead, a literature review was performed. Bioenergiportalen, published by Swedish JTI (Institutet för jordbruks- och miljöteknik), present a methane yield for liquid swine manure (8% DM) of 225 m^3/t DM (Bioenergiportalen, 2013). It is not described if this is laboratory scale data, or expected methane yields in full scale.

German KTBL (Kuratorium für Technik und Bauwesen in der Landwirtschaft) has a calculator for farm scale biogas plants (KTBL, 2013). The methane yield presented there for farm scale production of biogas from liquid swine manure (6% DM) is 202 m^3/t DM (or 252 m^3/t VS). In a Swedish literature review, a methane yield for liquid swine manure (8% DM) of 214 m^3 CH_4/t DM (268 m^3/t VS) is presented (Carlsson & Uldal, 2009). In the present study, a yield of 210 m^3 CH_4/t DM is selected.

For ley, the present study is based on experimental results performed by SLU and LU in both field trials and laboratory scale studies. The experimentally determined methane yields for samples harvested with a first cut early June and a second cut late August are summarized in Table A1. All 1st cut results (harvest 4-16 June) are pooled to one value, which is 290±18 m^3 CH_4/t DM. The second cut results (harvest 23-26 August) are in the same way pooled to a value of 245±9 m^3 CH_4/t DM. Practical experience from the trials have shown that the harvest time to achieve the high methane yields in early June is more difficult to pinpoint, reflected in the higher standard deviation achieved for 1st cut (June harvest). The VS as proportion of DM was not significantly different between samples, with a mean value of 91±1% of DM. This gives a VS-based methane yield for 1st and 2nd cut respectively of 318 and 269 m^3 CH_4/t VS.

Table A1. Summary of experimentally derived methane yield for different ley crop samples

Harvest date	Location	Methane yield[1] Mean with standard deviation m^3/ t DM	Fertilizer type in field trials
2010-06-04	Wrams Gunnarstorp	289 ± 7	digestate
2010-06-16	Wrams Gunnarstorp	273 ± 4	digestate
2008-06-10	Lönnstorp	298 ± 10	digestate
2008-06-16	Lönnstorp	287 ± 7	digestate
2008-06-10	Lönnstorp	307 ± 5	mineral fertilizer
2008-06-16	Lönnstorp	288 ± 9	mineral fertilizer
2008-08-26	Lönnstorp	240 ± 5	digestate
2008-08-23	Lönnstorp	245 ± 1	digestate
2008-08-26	Lönnstorp	240 ± 3	mineral fertilizer
2008-08-23	Lönnstorp	255 ± 7	mineral fertilizer

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

As a practical methane yield in the present study, 90% of the value achieved in laboratory scale will be used. Practical methane yields used in the study are summarized in Table A2.

Table A2. Methane yields used in the present study.

Feedstock	Methane yield m^3/t DM[1]	Uncertainty range
Liquid manure market swine	210	200-220
Grass/clover ley June harvest	261	245-277
Grass/clover ley August harvest	221	212-228

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

A.1.2 Carbon content

Determination of carbon content in the digestate was calculated based on feedstock values and model calculations. For ley, carbon content was analyzed in samples from 1st (n=4) and 2nd (n=4) harvest in cultivation trials in Lönnstorp (Table A1), giving a carbon content of 45.1±0.1% of DM for 1st harvest, and 45.5±0.3% of DM for 2nd harvest. These values are not significantly different, and a mean value of 45.3% based on all analyzed samples is used. For pig manure, data on carbon content has been given as 37% of DM for fattening pigs (Huang et al., 2006). For the residues after anaerobic digestion, carbon content was calculated based on initial values and the losses of carbon in relation to DM through biogas production as described above. The calculated digestate carbon content is shown in Table A4.

A.1.2 Maximum methane potential, B_0

The yield of methane that can be theoretically extracted from a certain amount of VS, assigned B_0, is in the default values presented by IPCC (2006) given as 450 m^3 CH_4/t VS for market swine and breeding swine. This B_0 value is given together with a conversion factor between volume and weight of methane of 0.67 kg/m^3, which is true at 101 kPa and 19°C. However, it is common practice to give methane volumes as normalized under 101 kPa (=1 atm) and 0°C, which is the standard conditions for all other gas volumes given in the present study. The IPCC B_0 value given under standard conditions is instead 421 m^3 CH_4/t VS for market and breeding swine. In the applied calculation model this maximum theoretical methane will give a VS-reduction of 97.7%, so even if very high, it is not unrealistic in the applied model.

For ley, the experimentally determined laboratory scale methane yields for 1st and 2nd harvest ley presented above (Table A2) are used as B_0 values.

From the input value for B_0, the B_0 of the effluent was calculated by subtracting the amount of methane produced in the digester, and dividing it by the amount of VS remaining in the digestate after degradation. At the methane yields used (Table A2), the resulting B_0-values of the digestate for the different scenarios are shown in Table A4. For the scenario with non-digested manure, the normalized B_0-value from IPCC (2006) is used, 421 m^3/t VS.

A.2 CALCULATED CHANGES RELATED TO NUTRIENTS

The input values for the manure and the different samples of ley are summarized in Table A4. Values for ley are based on own results from field trials with different harvest times. The ley is assumed to be field dried to a DM content of 35% and then stored as silage in bunker silos with a 5% DM loss during ensiling. No nutrient losses are assumed to occur during drying or ensiling. Nutrient and organic matter content for pig manure is taken from Focus on Nutrients (Greppa Näringen, 2011). Values in Table A4 are as manure comes from stable. When the manure is not used for biogas production and is stored in open tanks with floating crust cover, a 10% rain water dilution is added and organic matter and nutrient values are recalculated. Table A5 shows the resulting concentrations in manure and in digestate for the different scenarios before and after losses during storage of manure/digestate are subtracted.

Table A3. Content of organic material and nutrients in feedstock.

	At harvest		After ensiling					
	DM (% of ww)	VS (% of ww)	DM (%)	VS (%)	NH_4-N (g/kg)	N-tot (g/kg)	K (g/kg)	P (g/kg)
Ley year 0, undersown in oat	20	18	34	31	0.02	8.2	5.0	0.9
Ley year 0, undersown in wheat	32	30	34	31	0.01	7.1	4.0	0.8
Ley yr 1 of 1 and 2 of 2, June harvest	20	18	34	31	0.02	8.6	5.2	0.9
Ley yr 1 of 1 and 2 of 2, August harvest	22	20	34	31	0.02	10.3	5.8	1.2
Ley yr 1 of 2, June harvest	21	20	34	32	0.02	7.9	4.8	0.8
Ley yr 1 of 2, August harvest	32	31	34	32	0.01	7.1	4.0	0.8
Liquid pig manure	8.0	6.5	-	-	2.7	3.9	1.8	1.2

Table A4. Content of organic material and nutrients in manure and digestate for the different scenarios.

Scenario	A	B	C1	C2	C3	A	B	C1	C2	C3
	Before storage					After storage losses of N and organic matter				
Amount (t/yr)	6 050	5 290	8 390	10 820	9 560	6 040	5 280	8 380	10 810	9 550
DM (% ww)	7.3	4.3	9.4	11.4	11.0	7.1	4.2	9.4	11.3	10.7
VS (% DM)	81	63	74	81	76	81	62	74	81	75
C (% DM)	37	32	45	46	46	37	32	45	46	46
B_0 (m^3/t VS)	421	407	154	136	130					
N-tot (kg/t)	3.6	4.1	6.8	7.3	7.2	3.4	4.1	6.8	7.2	7.1
NH_4-N (kg/t)	2.4	3.4	4.3	4.2	4.3	2.3	3.3	4.2	4.2	4.2
P (kg/t)	1.1	1.2	1.3	1.2	1.2	1.1	1.3	1.3	1.2	1.2
K (kg/t)	1.6	1.9	3.7	4.0	3.9	1.6	1.9	3.7	4.0	3.9

APPENDIX B. CROP PRODUCTION

The environmental impact of crop production was evaluated. GHG emissions due to biomass production occurring in the different scenarios in the form of carbon dioxide equivalents (CO_2-eq.) were calculated using LCA methodology. Important parameters such as fertilizer use and carbon dioxide equivalent emissions are commented upon below.

B.1 FERTILIZATION LEVELS

Nitrogen fertilization was assumed to follow actual crop-specific levels at the Wrams Gunnarstops farm. Fertilization levels for phosphorus (P) and potassium (K) were calculated according to recommendations from the Swedish board of Agriculture (SJV, 2011). Nutrient recycling is assumed in Scenario A (pig manure) and Scenarios B, C1, C2 and C3 (biogas digestate). In practice, manure and digestate would be spread on fields to fulfill e.g. the phosphorous requirements. Application of organic fertilizers is limited to 22 kg P/ha as a 5-year average. This limitation has been accounted for, but the application of manure and digestate was simplified as an average application over all 650 ha of the farm. The difference between the amount of applied nutrients (N, P, K) and the recommended levels was assumed to be applied by mineral fertilizer.

B.2 CARBON DIOXIDE EQUIVALENT (CO_2-EQ.) EMISSIONS

Use of material and machinery in different production operations was analyzed in order to calculate direct and indirect emissions of carbon dioxide equivalents (Gissén et al., 2013; Prade et al., 2012). Analysis included direct emissions of CO_2 eq. due to consumption of diesel and electricity, and indirect emissions due to use of materials for buildings, machinery, fertilizer production etc. Except for fertilizers, the CO_2-eq. emissions were calculated considering the primary energy input and specific emission factors (Table B1).

Table B1. Specific emissions of carbon dioxide equivalent of selected materials

Material	Unit	CO_2-eq. emissions
Diesel	[g CO_2 eq. MJ^{-1}]	72*
Electricity	[g CO_2 eq. MJ^{-1}]	11.2*
Fertilizer N	[kg CO_2 eq. kg^{-1}]	6.6*
Fertilizer P	[kg CO_2 eq. kg^{-1}]	2.9*
Fertilizer K	[kg CO_2 eq. kg^{-1}]	0.44*
Machinery	[g CO_2 eq. MJ^{-1}]	85*
Seeds	[g CO_2 eq. MJ^{-1}]	50*
Pesticides	[g CO_2 eq. MJ^{-1}]	65*
Plastics	[g CO_2 eq. MJ^{-1}]	72*
Buildings	[g CO_2 eq. MJ^{-1}]	140*
Liming agent	[kg CO_2 eq. MJ^{-1}]	3.8**

References: *Börjesson et al., 2010, **UoT, 2008

B.3 CULTIVATION OPERATION

Cultivation operations for individual crops with corresponding primary energy input and GHG emissions are summarized in Tables B2-B6.

Table B2. Production operations in one-year ley crop.

Parameter	Primary energy input		GHG emissions	
	[MJ ha^{-1}]	[MJ ha^{-1}]	[kg CO_2-eq. ha^{-1}]	[kg CO_2-eq. ha^{-1}]
Scenario	C1	C3	C1	C3
Material				
Fertilizer N	3618	3252	497	447
Fertilizer P	54	38	20	14
Fertilizer K	783	23	72	2
Seeds	299	288	15	14
Pesticides etc.	314	314	20	20
Liming	22	22	82	82
Operations				
Sowing	0	0	0	0
Rolling	0	0	0	0
Fertilizer spreading	79	79	6	6
Mowing grass 1	368	520	27	38
Mowing grass 2	663	663	48	48
Mowing grass 3	649	649	47	47
Harvest 1 (forage harvester)	451	704	39	57
Harvest 2 (forage harvester)	677	677	50	50
Harvest 3 (forage harvester)	738	738	54	54
Transport of 1st harvest	574	779	43	58
Transport of 2nd harvest	689	689	53	53
Transport of 3rd harvest	512	512	56	56
Compaction silo	614	614	45	45
Feed in biogas plant	560	560	41	41
Storage				
Concrete bunker silo	277	277	39	39
Plastic cover for ensiling	495	495	36	36
Manure/digestate spreading				
Loading (pumping)	2	2	2	2
Transport to fields	177	200	13	15
Storage in satellite storage	26	23	4	3
Loading (pumping)	2	2	2	2
Spreading	298	281	22	21
Cultivation & harvest	9012	8247	1000	901
Storage	1412	1409	123	123
Transport	2516	2744	211	227
Total	**12940**	**12401**	**1334**	**1251**
Machinery	1886	1968	107	105
Diesel	5167	5701	443	488
Materials	1433	1418	196	195
Fertilizer	4454	3313	589	463
Total	**12940**	**12401**	**1334**	**1251**

Table B3. Production operations in two-year ley crop.

Parameter	Primary energy input	GHG emissions
	[MJ ha^{-1}]	[kg CO_2-eq. ha^{-1}]
Scenario	**C2**	**C2**
Material		
Fertilizer N	10606	1458
Fertilizer P	309	114
Fertilizer K	1841	169
Seeds	205	10
Pesticides etc.	628	41
Liming	43	164
Operations		
Sowing	0	0
Rolling	0	0
Fertilizer spreading	158	10
Mowing grass 1	368	27
Mowing grass 2	927	67
Mowing grass 3	532	39
Mowing grass 4	663	48
Mowing grass 5	649	47
Harvest 1 (forage harvester)	451	39
Harvest 2 (forage harvester)	1014	74
Harvest 3 (forage harvester)	799	59
Harvest 4 (forage harvester)	1082	50
Harvest 5 (forage harvester)	1143	54
Transport of 1st harvest	574	43
Transport of 2nd harvest	962	72
Transport of 3rd harvest	562	60
Transport of 4th harvest	1559	53
Transport of 5th harvest	512	56
Compaction silo	614	45
Feed in biogas plant	560	41
Storage		
Concrete bunker silo	314	44
Plastic cover for ensiling	530	38
Manure/digestate spreading		
Loading (pumping)	6	5
Transport to fields	436	32
Storage in satellite storage	21	3
Loading (pumping)	6	5
Spreading	175	13
Cultivation & harvest	21593	2483
Storage	1479	130
Transport	5177	367
Total	**28248**	**2980**
Machinery	4680	162
Diesel	9072	777
Materials	1741	300
Fertilizer	12756	1741
Total	**28248**	**2980**

Table B4. Production operations in oats.

Parameter	Primary energy input [MJ ha^{-1}]				GHG emissions [kg CO_2-eq. ha^{-1}]			
Scenario	A	B	C1	C2	A	B	C1	C2
Material								
Fertilizer N	5983	5712	4378	3686	823	785	602	507
Fertilizer P	78	78	30	1	29	29	11	0
Fertilizer K	78	78	0	0	7	7	0	0
Seeds	1648	1612	1419	1323	82	81	71	66
Pesticides etc.	497	497	497	497	32	32	32	32
Liming	22	22	22	22	82	82	82	82
Operations								
Stubble treatment (cultivator)	272	272	272	272	20	20	20	20
Ploughing	716	716	716	716	52	52	52	52
Harrowing	399	399	399	399	30	30	30	30
Sowing	81	81	81	81	17	17	17	17
Rolling	178	178	178	178	19	19	19	19
Fertilizer spreading	107	107	107	107	10	10	10	10
Spraying	241	241	241	241	14	14	14	14
Combine harvest	621	621	621	621	46	46	46	46
Drying (ventilator electricity only)	62	62	62	62	0	0	0	0
Drying (heat production)	463	463	463	463	12	12	12	12
Transport to user (30 km)	324	324	324	324	24	24	24	24
Manure/digestate spreading								
Loading (pumping)	2	2	2	3	3	2	2	3
Transport to fields	131	116	177	224	10	9	13	16
Storage in satellite storage	42	42	26	21	6	6	4	3
Loading (pumping)	2	2	2	3	3	2	2	3
Spreading	342	349	298	270	26	26	22	20
Cultivation & harvest	11262	10961	9258	8414	1289	1251	1029	916
Storage	567	567	551	546	18	18	16	16
Transport	459	444	505	553	39	38	42	46
Total	**12289**	**11972**	**10314**	**9514**	**1347**	**1307**	**1087**	**977**
Machinery	1494	1498	1455	1432	76	76	72	70
Diesel	2447	2434	2488	2531	210	209	213	217
Materials	2208	2172	1963	1863	203	201	189	184
Fertilizer	6139	5867	4408	3687	858	821	613	507
Total	**12289**	**11972**	**10314**	**9514**	**1347**	**1307**	**1087**	**977**

Table B5. Production operations in winter oilseed rape.

Parameter	Primary energy input [MJ ha^{-1}]					GHG emissions [kg CO_2-eq. ha^{-1}]				
Scenario	A	B	C1	C2	C3	A	B	C1	C2	C3
Material										
Fertilizer N	6821	6587	5469	4723	4856	938	906	752	649	668
Fertilizer P	155	155	112	83	81	57	57	41	31	30
Fertilizer K	145	145	8	0	49	13	13	1	0	5
Seeds	86	84	76	71	72	4	4	4	4	4
Pesticides etc.	662	662	662	662	662	43	43	43	43	43
Liming	22	22	22	22	22	82	82	82	82	82
Operations										
Stubble treatment (cultivator)	272	272	272	272	272	20	20	20	20	20
Ploughing	716	716	716	716	716	52	52	52	52	52
Harrowing	598	598	598	598	598	45	45	45	45	45
Sowing	81	81	81	81	81	17	17	17	17	17
Rolling	178	178	178	178	178	19	19	19	19	19
Fertilizer spreading	107	107	107	107	107	10	10	10	10	10
Spraying	213	213	213	213	213	13	13	13	13	13
Combine harvest	688	688	688	688	688	51	51	51	51	51
Drying (ventilator electricity only)	112	112	112	112	112	1	1	1	1	1
Drying (heat production)	831	831	831	831	831	21	21	21	21	21
Transport to user (30 km)	356	356	356	356	356	27	27	27	27	27
Manure/digestate spreading										
Loading (pumping)	2	2	3	3	3	3	2	3	3	3
Transport to fields	131	116	212	276	262	10	9	15	20	19
Storage in satellite storage	42	42	27	22	22	6	6	4	3	3
Loading (pumping)	2	2	3	3	3	3	2	3	3	3
Spreading	342	349	284	248	254	26	26	21	19	19
Cultivation & harvest	11085	10856	9486	8662	8850	1390	1359	1171	1055	1077
Storage	985	984	970	964	964	28	28	26	25	25
Transport	491	475	573	639	624	42	40	47	53	51
Total	**12561**	**12316**	**11028**	**10265**	**10437**	**1460**	**1426**	**1244**	**1132**	**1153**
Machinery	1977	1981	1928	1899	1904	89	89	84	82	82
Diesel	2652	2638	2725	2783	2770	227	226	233	238	237
Materials	811	809	786	776	777	135	135	133	132	132
Fertilizer	7121	6887	5589	4806	4987	1008	976	794	680	702
Total	**12561**	**12316**	**11028**	**10265**	**10437**	**1460**	**1426**	**1244**	**1132**	**1153**

Table B6. Production operations in winter wheat.

Parameter	Primary energy input [MJ ha^{-1}]					GHG emissions [kg CO_2-eq. ha^{-1}]				
Scenario	A	B	C1	C2	C3	A	B	C1	C2	C3
Material										
Fertilizer N	9180	8909	7575	6883	7210	1262	1225	1042	946	991
Fertilizer P	170	170	121	93	105	62	62	45	34	39
Fertilizer K	155	155	1	0	64	14	14	0	0	6
Seeds	1710	1686	1554	1492	1527	85	84	78	75	76
Pesticides etc.	662	662	662	662	662	43	43	43	43	43
Liming	22	22	22	22	22	82	82	82	82	82
Operations										
Stubble treatment (cultivator)	272	272	272	272	272	20	20	20	20	20
Ploughing	716	716	716	716	716	52	52	52	52	52
Harrowing	598	598	598	598	598	45	45	45	45	45
Sowing	81	81	81	81	81	17	17	17	17	17
Rolling	178	178	178	178	178	19	19	19	19	19
Fertilizer spreading	107	107	107	107	107	10	10	10	10	10
Spraying	284	284	284	284	284	15	15	15	15	15
Combine harvest	944	944	944	944	944	69	69	69	69	69
Drying (ventilator electricity only)	259	259	259	259	259	1	1	1	1	1
Drying (heat production)	1940	1940	1940	1940	1940	51	51	51	51	51
Transport to user (30 km)	479	479	479	479	479	35	35	35	35	35
Manure/digestate spreading										
Loading (pumping)	2	2	2	3	2	3	2	2	3	2
Transport to fields	131	116	177	224	200	10	9	13	16	15
Storage in satellite storage	42	42	26	21	23	6	6	4	3	3
Loading (pumping)	2	2	2	3	2	3	2	2	3	2
Spreading	342	349	298	270	281	26	26	22	20	21
Cultivation & harvest	15419	15131	13413	12601	13050	1823	1785	1559	1448	1506
Storage	2241	2241	2225	2220	2222	59	59	56	56	56
Transport	615	599	660	709	683	50	49	53	57	55
Total	**18275**	**17971**	**16298**	**15530**	**15955**	**1932**	**1892**	**1668**	**1560**	**1616**
Machinery	3342	3347	3303	3280	3288	120	120	116	114	114
Diesel	2993	2980	3034	3077	3055	256	255	260	264	262
Materials	2435	2412	2263	2196	2233	217	215	207	203	205
Fertilizer	9504	9233	7697	6976	7379	1339	1301	1086	980	1036
Total	**18275**	**17971**	**16298**	**15530**	**15955**	**1932**	**1892**	**1668**	**1560**	**1616**

APPENDIX C. NITROGEN-RELATED EMISSIONS

Losses of nitrogen with impact on climate occur in the form of biogenic nitrous oxide (N_2O) emissions, of evaporation (volatilization) of ammonia (NH_3) and of leakage of nitrogen containing compounds to water. In this appendix, the background to the emissions is briefly introduced, and the background data for calculation of emissions are presented.

N_2O is a potent greenhouse gas. Expressed as global warming potential (GWP) it is converted to CO_2-eq., and in a 100 year perspective 1g N_2O corresponds to 296g CO_2-eq. (IPCC, 2006). It can be formed and emitted from chemical processes, like e.g. at the production of mineral fertilizers (see Appendix E), but the term biogenic refers to the fact that the N_2O is produced by naturally occurring microorganisms. The production occurs through the processes of nitrification and denitrification. Nitrification is an aerobic two-step process where ammonium (NH_4^+) via nitrite (NO_2^-) is oxidized to nitrate (NO_3^-). The growth of nitrifying microorganisms is strongly influenced by temperature, and depending on the presence of oxygen. Denitrification is a reduction process, so normally occurs in the absence of oxygen, where nitrate is reduced via nitrite to nitrous oxide (N_2O) or nitrogen gas (N_2). The N_2O is an intermediate in this degradation, and seems to be formed in favor of nitrogen gas when oxygen is present in low concentrations (Takaya et al., 2003). The risk for these conditions has been shown to prevail when the soil moisture content is higher, while soil temperature has less effect (Maag & Vinther, 1996). The presence of an easily degradable carbon source is required for denitrification to occur, and even at high nitrogen concentrations in soil, denitrification rates have been shown to be low in the absence of an easily available carbon source (Weier et al., 1993). Soil pH has been shown to be a main driver for the ratio of N_2O/N_2 production in denitrification, where N_2O is the main product under acidic conditions (Giles et al., 2012; van den Heuvel et al., 2011). The biogenic N_2O formation will thus occur in environments where;

- Nitrogen in different forms is available
- Both aerobic (oxygen is present) and anoxic (no oxygen is present, but other electron acceptors like nitrite (NO_2^-) and nitrate (NO_3^-) are available) conditions occur
- Easy degradable organic compounds are present

Examples of this kind of environment that are relevant in relation to the present study are managed soils, where both added nitrogen in the form of mineral fertilizer and biofertilizers, and nitrogen containing crop residues generate N_2O emissions. Also, manure management systems and digestate handling will cause emissions. The basis for the calculations of N_2O emissions are in the following text presented as emission factors, percentages which are based on the N-tot content of different types in different parts of the systems studied.

NH_3 is present in both manure and digestate, and then both as the volatile form (NH_3, ammonia) and the protonated form ammonium (NH_4^+). The major factor that determines which form that is dominating is the pH, and NH_4^+ dominates at pH in the range of 7-8, which is the case for manure and digestate. Ammonium-nitrogen (NH_4-N) is thus used in this study to refer to the mineralized form of nitrogen, in contrast to organically bound nitrogen (N-org) or total nitrogen (N-tot). Between 1-10 % of the NH_4-N in the manure and the digestate will though be present as NH_3, which is volatile. The share increases with increasing pH, which is why the risk for NH_3 losses increases with digestion of manure, when the pH normally rises. The share on N-tot that is mineralized to NH_4-N also increases during digestion, as also shown in the present study (Clemens et al., 2006;

Wulf et al., 2002). Volatilized NH_3 gives effects on eutrophication and acidification, which are environmental impacts that have not been quantified in the present study. NH_3 is not a greenhouse gas in itself, but it is assumed to reach other ecosystems, where it gives indirect emissions of biogenic N_2O. The calculation of the indirect emission is based on the IPCC default value, where 1% of the N lost as NH_3 is assumed to be converted to N_2O in other ecosystems (IPCC, 2006). The emission of NH_3 is however specific for different situations.

Leakage of nitrogen to water occurs mainly in the form of the ion nitrate (NO_3^-). The average nitrogen leakage to water from Swedish farmland was estimated to 18 kg N/ha based on conditions in 2005 (Johnsson et al., 2008). The difference between regions and soil types is large, with a variation of 5-47 kg N/ha. The difference between crops within one region also varies. Nitrogen leakage also varies for land only fertilized with mineral fertilizer or with manure with addition of mineral fertilizer, where the leakage is 0-10 kg N/ha higher for the latter. 0-10 kg N/ha should thus be attributed to the biofertilizer addition. In the present study, regional data on hectare related nitrogen leakage is used, and is increased when manure or digestate utilization is increased. The higher fraction of easily available nitrogen in digestate has been shown to give a faster nitrogen uptake in the crops, which would give a lower leakage. In the present study, however, manure and digestate is treated as equal giving equal emissions regarding this aspect.

Leaching of NO_3^- gives effect on eutrophication, which is an environmental impact not quantified in the present study. NO_3^- is assumed to reach other ecosystems, where it gives indirect emissions of biogenic N_2O. The calculation of the indirect emission is based on the IPCC default value, where 0.75% of the N lost through leakage is assumed to be converted to N_2O in other ecosystems (IPCC, 2006). The leakage of N is however specific for different scenarios.

C.1 STORAGE OF MANURE AND DIGESTATE

C.1.1 Direct N_2O emissions

Nitrous oxide (N_2O) can be released through a direct emission from manure during storage. IPCC present values for different manure handling systems, which also are used in the Swedish national inventory report (IPCC, 2006; Naturvårdsverket, 2013). For liquid manure with a natural crust cover, the emission factor (EF) is given as 0.5% of the N-tot excreted by the animal. For liquid manure without crust cover, and for manure management through anaerobic digestion, the direct N_2O emission factor is given as zero (IPCC, 2006b). A pilot scale study where non-digested cattle manure was stored under natural crust cover, and digested manure was stored with addition of straw (to create an artificial crust cover) and covered with a roof gave both on average emissions that have been recalculated to an EF of 1.4% (Clemens et al., 2006). For swine manure and liquid storage, a Swedish study has shown an annual average EF with floating crust cover of 0.7% of N-tot, and no emissions when storage occurred without crust cover or with plastic cover (Rodhe et al., 2012). In the present study, the IPCC EF for manure with crust cover is applied with no emissions for the roof covered digestate storages.

C.1.2 Indirect emissions

Indirect N_2O emissions will occur due to the volatilization of NH_3 during storage (IPCC, 2006). These losses are much influenced by manure type, handling method (liquid or solid) and for liquid manure different types of cover. For liquid manure, a floating crust will reduce emissions by 50-60% compared to no cover, and roof cover will give 90-95% lower losses (Jordbruksverket, 2007). A review by Karlsson and Rodhe is presented by the Swedish EPA in the national inventory report, and their data is used for estimating NH_3-losses (Karlsson & Rodhe, 2002). The assumptions applied in the present study are that emissions from crust covered storage of liquid swine manure when storage is filled from below are 4% (share of N-tot) (Naturvårdsverket, 2013).The digestate, which is stored under roof cover, emits 1%. The risk for ammonia volatilization is higher for digested manure due to the higher concentration of ammonia and the increase in pH compared to untreated manure. Thus, roof cover, as at the full scale plant supplying data for the present evaluation, can be important to reduce emissions for digested manure.

C.2 MANAGED SOILS

C.2.1 Direct N2O emissions

According to IPCC, the main determinator for N_2O emissions from managed soils is the content of inorganic nitrogen in the soil (IPCC, 2006). A high content of available nitrogen increases the rate of nitrification and denitrification. To quantify the addition of nitrogen is thus the most relevant input parameter in the calculations (IPCC, 2006). The actual N_2O emissions from managed agricultural soils will vary much depending on factors like soil type, soil pH, water content, climate, land use etc. These variations are however not considered in the calculations because limited data is available to support more emission factors. The IPCC guidelines provide general facts about the calculations, and general emissions factors that should be used if not better national or regional level data exists. If regional/national data are to be used, they need to be rigorously documented (IPCC, 2006). In the following, the IPCC recommendations are discussed in relation to other data available, and if not country-specific data is available of if it is poorly documented, general IPCC data is used.

The biogenic N_2O emissions are divided into direct and indirect emissions. Direct emissions are related directly to the nitrogen present in the soil, using emission factors (EF) expressed as kg N_2O-N/kg N-tot. Nitrogen sources and amounts are quantified, and different EF are then used to convert nitrogen added to the soil in different forms to N_2O emissions.

The sources for N input in this study are;

- Mineral fertilizer
- Biofertilizer in the form of manure
- Biofertilizer in the form of digestate
- Crop residues
- Nitrogen released or taken up by the degradation or formation of soil organic matter.

In the IPCC guidelines, the same general EF is suggested for all types of N input (IPCC, 2006). According to the IPCC guidelines, the given EF has been calculated based on the total addition of

N in the field, so before subtraction of losses due to volatilization and leaching/runoff (IPCC, 2006). The EF given as a default value for all the above categories by IPCC is in the update of 2006 is 1%, where the given uncertainty range is 0.3%-3%. However, in the National Inventory Report for Sweden, the Swedish EPA have presented country specific EFs for some types of N input (Naturvårdsverket, 2013). These country specific EFs are based on a literature review carried out in 2001 where studies performed in countries with comparable management and climatic conditions (Sweden, Canada and northern Europe) are summarized (Kasimir Klemedtsson, 2001). Emissions of N_2O-N from addition of mineral fertilizer on mineral soils were found to be 0-0.8% of added nitrogen, and an EF of 0.8% is used in the national inventory report (Naturvårdsverket, 2013). For fertilization with animal manure, emissions of N_2O-N range between 0.6% and 8% of added N (Kasimir Klemedtsson, 2001). The country-specific EF suggested and applied by the Swedish EPA based on these data and a regression model is 2.5% of N-tot added, which is used as the base case value in the present study (Naturvårdsverket, 2013). In the sensitivity analysis, applying the IPCC default value also for manure (1%) is also investigated. For swine manure, a Swedish field study has shown EF of 0.5% to 1.4% depending on spreading technique and time, where soil moisture had significant influence on emissions (Rodhe et al., 2012). Emissions can thus be lower that the applied national EF, and in the sensitivity analysis an EF of the average value 0.9% based on the above study is evaluated and compared to national emission data for the digestate (as later presented).

For crop residues, the IPCC default value is used. The amounts of crop residues based on different calculation models are varied, as presented in section 8.1, thus varying this contribution. The content of N in the crop residues is however kept fixed. The values used are taken from IPCC guidelines (IPCC, 2006b).

Table C1. Nitrogen content in crop residues.

Crop	*Nitrogen content % of DM*	
	Above ground	Below ground
Ley	2.5	1.6
Winter rape	0.8	0.9
Oat	0.7	0.8
Winter wheat	0.6	0.9

Biogenic N_2O emissions from organic N-additions like compost, digestate from sewage treatment etc. is according to IPCC to be included using the same default value as for all N-additions (IPCC, 2006b). There is, according to Swedish EPA, no Swedish research that will motivate a national EF concerning N_2O emissions from sewage sludge (Naturvårdsverket, 2013). In the present study, the IPCC default EF of 1% of added N for compost and sewage sludge is also used for digested manure. Even though the concentration of ammonia is higher than in non-digested manure, the easily degradable carbon has been converted to biogas, and only the more recalcitrant organic compounds will remain in digested manure. The presence of an easily degradable carbon source is, however, required for denitrification to occur, and if absent, denitrification rates have been shown to be low even at high nitrogen concentrations in soil (Weier et al., 1993). A lower EF for digestate is thus motivated. A lower N_2O emission after soil application of digested cattle manure (EF for N-tot added 0.07% winter and 0.44% summer) compared to non-digested manure (EF 0.19% winter and

0.59% summer) has also been shown in experimental field studies under Swedish conditions (Rodhe et al., 2013). In lack of data specific for digested pig manure, the mean value for cattle manure digestate, which as an annual average has been calculated to 0.2%, is used in the sensitivity analysis.

If soil carbon is degraded, the related mineralized N is also released (8-12% of C) and should be included in the calculation of direct N_2O emissions. In the same way, N is taken up and made unavailable for biogenic N_2O formation if soil organic matter levels increase. For the region investigated in the present study (Götalands södra slättbygder, Gss), soil carbon levels have been predicted as generally stable (Andrén et al., 2008), and N contributions from degradation of soil organic matter is not generally included in calculations of N_2O emissions. In the present study, however, the soil carbon increase or decrease is a central part of the study, and the related N release or uptake is included in the calculations of N_2O emissions. The C:N ratio of soil organic matter is assumed to be 10:1, and the N_2O emissions of this nitrogen removal or contribution is calculated with and EF of 1% of N taken up or released. The EF applied for each type of nitrogen source is summarized in Table C2.

Table C2. Emission factors (EF) for different types of N addition to managed soils (kg N_2O-N/kg N). The EF is based on all N added to the soil.

Source of nitrogen	*EF*	*Comment*
Mineral fertilizer	0.8%	National EF
Pig manure	2.5%	National EF
• sensitivity analysis	1%%	IPCC default
• sensitivity analysis	0.9%	National experimental
Crop residues	1.0%	IPCC default
N from soil organic matter	1.0%	IPCC default
Digestate	1.0%	IPCC default
• sensitivity analysis	0.2%	National experimental

C.2.2 Indirect emissions

Indirect emissions include N that is volatilized from managed soils, that is deposited and converted to N_2O in other ecosystems. The N contributing to these emissions are the mineral fertilizers and the organic and inorganic N added through organic fertilizers. The share of N assumed to be volatilized is different for these fractions. Volatilization of N to air for mineral fertilizer is by IPCC given as 10%, but that is based on fertilization with urea, which is used very little in Sweden. In Denmark, an inventory of N volatilization as ammonia has been performed (Hutchings et al., 2001), where volatilization was measured as 2% at mineral fertilization. The Swedish EPA, present thorough statistics on types of fertilizers sold in Sweden, and a volatilization factor based on the actual product mix is used. With ammonium nitrate dominating, which has an emission factor of 0.9%, the total weighted emission factor for the mineral fertilizer mix in 2010 presented as 0.91%, which is used in the present study (Naturvårdsverket, 2013).

For calculations of loss of NH_3 to air at manure fertilization, emissions to air may vary much, between 3-70% of added N, and depend on manure handling system (solid or liquid), spreading method, spreading time and time for mulching (Naturvårdsverket, 2013). The Swedish EPA present country specific data on these emissions based on a study by Karlsson and Rodhe (2002). The manure or digestate is in all scenarios in the present study spread in spring, and in growing crops. This argues for a high degree of volatilization, and a loss of 30% of NH_4-N added is used for spreading in ley, 15% for the other crops. Since the fraction of NH_4-N is higher in digestate compared to manure, this will result in higher emissions of NH_3 when digestate is used.

A Danish study on undigested and digested liquid cattle manure spread on silty loam soils with winter cereals has given an emission of about 13% of added NH_4-N for manure, compared to approximately 14.5% for digested manure. Both values are in good agreement with the value of 15% chosen in the present study (Möller & Stinner, 2009). A Swedish study with spring application directly followed by harrowing and sowing has shown emissions that vary much for non-digested and digested cattle manure, where digested manure emitted 30% of added NH_4-N as NH_3-N while non-digested only 4% (Rodhe et al., 2013). This was however in a field with no vegetation and before sowing, with harrowing after 4 hours.

Ammonia emissions to air from crops and crop residues do occur, NH_3 emission from beet tops and green manuring crops have been seen to be as high as 5 kg N/ha in Denmark (Hutchings et al., 2001), but is not included in the IPCC model. We also exclude it in the present study since we do not let green manuring crops stay on the ground.

The second category for indirect emissions of N_2O is leakage to water. Nitrogen leakage to water is reported in national estimates by the Swedish EPA (Johnsson et al., 2008). The data is calculated through a model that has been validated with leaching experiments with good results, and is used in the national GHG inventories (Naturvårdsverket, 2013).The data is reported region by region for different soil types and land use. The leakage is given both for land fertilized with mineral fertilizer and with biofertilizer plus mineral fertilizer addition for all the main crops in the region. The region of Gss is divided in two production areas (1a, Skånedelen and 1b, Hallandsdelen), where average leakage for region 1a was 32 kg N/ha and for 1b 46 kg N/ha. The average N leaching for Sweden was 18 kg N/ha for both 2005 and 2009. The region 1a is further divided in soil types, where sandy loam represents 72% of the land in the region. Crop specific data on N leakage for sandy loam for region 1a (Skånedelen av Gss) is used, and the values are presented in Table C3 (Johnsson et al., 2008).

Table C3. Nitrogen leakage for the Skåne part of the Gss-region. Data depending on fertilization strategy and crop for sandy loam, which represents 72% of the agricultural land in this region.

Fertilizer addition	*Nitrogen leakage (kgN/ha)**			
	Winter wheat	**Ley***	**Winter oil seed rape**	**Oat**
Mineral fertilizer	34	14	55	44
Manure + mineral fertilizer	40	16	60	43

* for each year of a ley renewed after 3 years. In the present study, 1 and 2 year ley is investigated. The total leakage over the three year period is then instead shared between 1 or 2 years.

Biofertilized areas have higher leakage/runoff than areas only fertilized with mineral fertilizer, but no difference is made depending on if the biofertilizer is digested or not. Digested manure has a higher content of mineralized nitrogen (ammonia), which is more readily taken up by the crops, and is in this study always spread in growing crops. It could thus be argued that leakage should be lower when digestate is applied. No such difference is thus made. In the different scenarios, different amounts of manure or biofertilizer are available, and the share of land using only mineral nitrogen or a mix of biofertilizer and mineral nitrogen will vary between the scenarios depending on available amounts.

APPENDIX D. METHANE EMISSIONS

D.1 STORAGE OF MANURE AND DIGESTATE

The IPCC model according to Tier 2 will be used for the calculations, but has been simplified as shown in Eq. 1 (IPCC, 2006b) since amounts of VS can be carefully determined, and only one temperature zone is relevant for one specific biogas plant or manure storage facility.

$CH_4 = VS*B_0*0.67\ kg/m^3*MCF$ (Eq. 1)

Where:
CH_4 = emissions of methane (kg CH_4)
VS = volatile solids in manure (kg)
B_0 = maximum methane producing capacity (m^3 CH_4/kg VS)
MCF = Methane conversion factor (%)

The conversion factor between volume and weight of CH_4 is by IPCC given as 0.67 kg/m^3, which is true at 101 kPa and 19°C. However, it is common practice to give CH_4 volumes as normalized under 101 kPa and 0°C, conversion factor is then 0.72 kg CH_4/m^3. The IPCC B_0 values are however given together with the 0.67 kg/m^3 conversion factor, so that factor is used together with these B_0-values.

B_0 is the yield of methane that can be theoretically extracted from a certain amount of VS. In the default values presented by IPCC (2006), B_0 is set to 450 m^3 CH_4/t VS for market swine and breeding swine. These B_0 values are also applied by the Swedish EPA (Naturvårdsverket, 2013).

The MCF depends on factors such as manure type, temperature and storage time. IPCC suggest MCF as 10% for liquid manure at annual average temperatures of 10°C. For liquid manure, these values are if the storage tank has a natural crust cover. Without crust cover, the default value is higher, 17%. The uncertainty in these emission factors is given as ±30%. The value for liquid manure is based on a Danish study (Sommer et al., 2000) and an American publication (Mangino et al., 2002) on liquid swine manure handling. Under Swedish conditions, the MCF has, based on new measurements presented by Rodhe et al. (2008) been set to 3.5% in the latest national inventory report (Naturvårdsverket, 2013), which is also the value applied in the base case in the present study.

For the digestate from pig manure or a mix of pig manure and ley, the B_0-values are calculated as presented in Appendix A. In the base case, the same MCF is applied for digestate as for manure, 3.5%.

In a sensitivity analysis, data from Swedish studies are used. For liquid pig manure with a floating crust, CH_4 emissions corresponding to an average MCF of 2.8% has been presented (Rodhe et al., 2012). For digestate from cow manure digestion, high CH_4 emissions has been measured for digested cow manure, with annual average MCF of 12%. This value is applied for digestate from pig manure and for co-digested manure and ley in the sensitivity analysis (Rodhe et al., 2013).

Full scale data from monitoring of farm scale biogas plant has presented data on maximum CH_4 production from digestate (FNR, 2010). The study presents data from 61 biogas plants digesting varying share of energy crops and manure. Data from the 13 plants with highest share of manure (56-90%, average 70% of feedstock wet weight) show a residual methane potential for the digestate of on average 2.3 m^3 CH_4/t digestate. This is a methane potential achieved in laboratory scale digestion at 20-22°C during 60 days, so can be considered the maximum theoretical production at this temperature. Given as a share of the methane produced in the biogas plant, this represents 4.6% of the methane collected in the biogas process. In another German study, the measured leakage of CH_4 from digestate storage related to the actual electrical output during time of measurement was presented as 6.25 g CH4/kWh for six farm scale biogas plants. With an assumed electric efficiency of 40%, this leakage amounted to average 3.5% of the utilized CH_4, with a range of 0.2-10.2% (Liebetrau et al., 2013). It was pointed out that these results do not represent average emission over a longer period of time, and that the large variability in data could not be explained within the frames of the study. The investigated plants were digesting energy crops in co-digestion with 0-94% manure.

APPENDIX E. SOIL CARBON MODELLING

In order to calculate changes in the soil carbon content as influenced by the choice of crop rotation, the well-described and well-applied Introductory Soil Carbon Balance Model (ICBM) was used (Andrén & Kätterer, 1997; Kätterer & Andrén, 2001). The model was applied to calculate the soil carbon content according to carbon inputs and mineralization rates. For this purpose, the model was modified to account for different input types with specific humification factors. Prior to analysis of the project scenarios, the model was calibrated with data derived from a close-by long term soil carbon field trial.

E.1 MODEL

Figure E1 shows the general ICBM structure. For this study we choose to account for three input types, i.e. above-ground biomass, below-ground biomass and added biomass (manure/digestate).

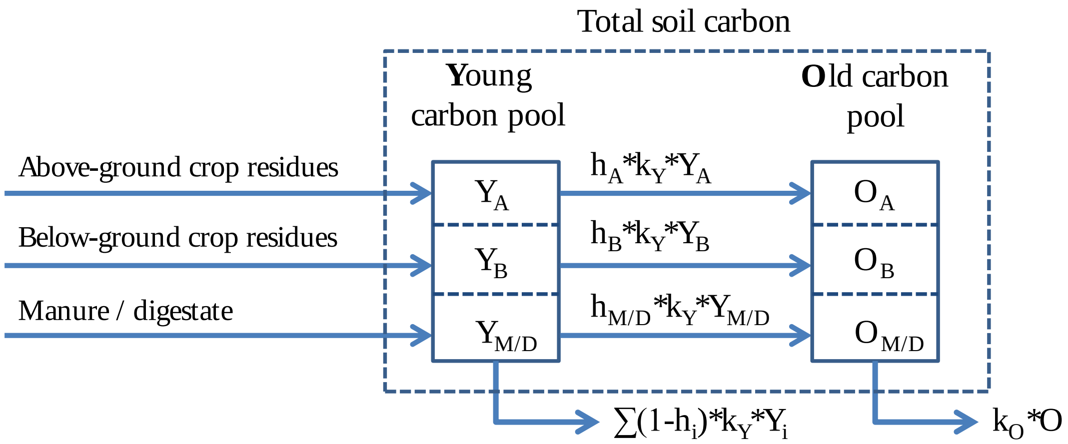


Figure E1. ICBM outline including the different biomass inputs, adjusted from Andrén and Kätterer (1997). Y_i = young carbon pools of different origin; O_i old carbon pools of different origin; h_i humification factors for different residues; k_i reaction coefficients for the different carbon pools.

All carbon from crop residues and biomass additions enters the young carbon pool (Y). This pool has an outflow of carbon with a relatively high reaction coefficient of k_Y=0.8 (i.e. within one year 1-exp(-0.8)=55% of the carbon leaves the young carbon pool again (Andrén & Kätterer, 1997). From here, only a fraction described by a humification factor (h) enters the old carbon pool, which has a much lower reaction coefficient (k_O) than the young carbon pool. The output from the old carbon pool describes losses from the soil carbon by mineralization.

E.2 INPUTS

Inputs to the young soil carbon pool can come from crop residues or added biomass, e.g. manure. Crop residues usually comprise aboveground biomass such as stubble and straw (if left in the field) and belowground biomass such as root and root exudates ("extra-root biomass")(Bolinder et al., 2007). Two different data sources for calculation were applied in parallel and compared: (a) IPCC data and (b) data from Nordic studies. Crop residue amounts were calculated for each crop starting with the harvested yield of the desired crop part, e.g. grains, beets, seeds and ley crop biomass.

IPCC data given as a slope and intercept (ley crops: slope only) was used to calculate the above-ground crop residue dry matter from the harvested biomass yield, Table E1 (IPCC, 2006). The below-ground crop residue dry matter was then calculated according to a ratio of below-ground to above-ground biomass. With Nordic data, above-ground crop residue amounts for cereals and rape-seed were calculated using straw-to-kernel ratios Table E3.

Table E1. IPCC data for calculation of crop residue amounts (IPCC 2006)

Crop	Slope	Intercept [t DM ha-1]	Below-ground to above-ground ratio
Winter wheat	1,61	0,40	0,23
Sugarbeet	0,10	1,06	0,20
Ley crops	0,30	0,00	0,80
Spring barley	0,98	0,59	0,22
Spring rapeseed	1,09	0,88	0,22
White mustard	1,09	0,88	0,22
Oats	0,91	0,89	0,25
Winter rapeseed	1,09	0,88	0,22

Table E2. Data used for model parameterization (Kätterer et al., 2011)

Crops	Humification coefficient h		
	Above-ground residues	Below-ground residues	Added biomass
Winter wheat	0,150	0,350	
Sugarbeet	0,120	0,350	
Ley crops	0,120	0,350	
Spring barley	0,150	0,350	
Spring rapeseed	0,150	0,350	
White mustard	0,120	0,350	
Oats	0,150	0,350	
Winter rapeseed	0,150	0,350	
Manure			0,270
Digestate			0,410

Straw recovery rates were used in cases where straw was removed from the field. For sugar beets, a shoot-to-root ratio was used to calculate the amount of above-ground residues. For ley crops, the amount of above-ground residues (stubble) was calculated from the biomass yield and a recovery coefficient (own unpublished data). 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).

The final resulting amounts of crop residues used in the scenario calculations are given in Table E4 at the end of this chapter.

E.3 PARAMETERIZATION

The model was parameterized using Nordic literature data, se Table E2.

E.4 CALIBRATION

In order to adapt the ICBM to Nordic conditions, the model was calibrated against data derived from the long-term soil carbon field experiment in Ekebo, Sweden (Kirchmann et al., 1999). The Ekebo soil carbon field experiment includes two different crop rotations. One was designed as a crop rotation for an animal production farm, with all cereal straw and sugar beet tops removed. The other crop rotation was designed for a pure plant production farm, with all straw and sugar beet tops left in the field. For each rotation 16 different fertilization regimes (all combinations of 4 nitrogen and 4 phosphorous/potassium fertilization levels) were tested. The experiment started 1957 and is ongoing with regular soil carbon content analyses.

Calibration was done using the reaction coefficient of the old carbon pool (k_O) as a variable to fit model soil carbon predictions to the measured soil carbon data. This was done using crop residue data as computed by (a) the IPCC calculation and (b) by the Nordic calculation for comparison. The prediction power of the model was computed by maximizing the coefficient of determination (R^2) of the measured and predicted data.

Table E3. Nordic data for calculation of crop residue amounts.

Crop	Recovered yield assumed in analysis	Grains, seeds, beets	Straw/ley			Stubble/litter		Roots	Extra root	Crop residue input		References
			in the field	recovery						above-ground	below-ground	
	[t DM ha^{-1}]	[rel. yield]	[rel. yield]	[%]	[rel. yield]	[%]	[rel. yield]	[rel. yield]	[rel. yield]	[rel. yield]	[rel. yield]	
Winter wheat	6,5	1,00	0,57	75	0,43	25	0,14	0,31	0,20	1,24	1,21[a]	(Bolinder et al., 2007; Kätterer et al., 2011; Nilsson & Bernesson, 2009)
Sugarbeet		1,00	0,30				0,00	0,01	0,01	0,30	0,02	(Bolinder et al., 2007; Gissén et al.; Koga et al., 2011; Kätterer et al., 2011)
Ley crops, 0 year after oat	1,5		1,25	80	1,00		0,00	0,00	0,72	0,00	0,72	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 0 year after wheat	2,5		1,25	80	1,00		0,00	0,00	0,72	0,00	0,72	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 1/1 year	9,0		1,25	80	1,00	20	0,25	1,11	0,72	0,25	1,83	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 1/2 year	12,0		1,25	80	1,00		0,00	0,00	0,72	0,00	0,72	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 2/2 year	9,0		1,25	80	1,00	20	0,25	1,27	0,83	0,25	2,10	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Spring barley		1,00	0,35	0	0,00	0	0,00	0,32	0,21	0,35	0,53	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Spring rapeseed		1,00	0,90	85	0,76	15	0,13	0,31	0,20	0,90	0,52	(Bolinder et al., 2007; Nilsson & Bernesson, 2009; Pietola & Alakukku, 2005)
White mustard		1,00	0,67	0	0,00	0	0,00	0,51	0,33	0,67	0,83	(Akhtar & Mashkoor Alam, 1992; Arp et al., 2010; Bolinder et al., 2007)
Oats	4,0	1,00	0,50	65	0,32	35	0,17	0,43	0,28	1,54	2,18[b]	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Winter rapeseed	2,5	1,00	0,92	85	0,78	15	0,14	0,23	0,15	1,18	0,49	(Becka et al., 2004; Nilsson & Bernesson, 2009)

[a] *Only wheat. In scenario C3, if the undersown ley crop biomass is included the value is 2.62.*

[b] *Only oats. In scenarios C1 and C2, if the undersown ley crop biomass is included the value is 3.30*

Table E4. Amounts of crop residues used in the scenario calculations.

Crop	Recovered yield assumed in analysis	Straw/ley			Stubble/litter		Roots	Extra root	Crop residue input		References
		in the field	recovery						above-ground	below-ground	
	[t DM ha^{-1}]	[t DM ha^{-1}]	[%]	[t DM ha^{-1}]	[%]	[t DM ha^{-1}]	[t DM ha^{-1}]	[t DM ha^{-1}]	[t DM ha^{-1}]	[t DM ha^{-1}]	
IPCC											
Winter wheat	6,5	10,87	75	8,15	25	2,72	3,99	0,00	10,07 a	3,99	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Ley crops, 0 year after oat	1,5	1,95	80	1,50		0,00	0,00	0,00	0,00	0,00	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Ley crops, 0 year after wheat	2,5	3,25	80	2,50		0,00	0,00	0,00	0,00	0,00	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Ley crops, 1/1 year	9,0	11,70	80	9,00	20	2,70	9,36	0,00	2,70	9,36	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Ley crops, 1/2 year	12,0	15,60	80	12,00		0,00	0,00	0,00	0,00	0,00	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Ley crops, 2/2 year	9,0	11,70	80	9,00	20	2,70	9,36	0,00	2,70	9,36	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Oats	4,0	4,53	65	2,94	35	1,59	2,13	0,00	4,53	2,13	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Winter rapeseed	2,5	3,61	85	3,06	15	0,54	1,34	0,00	3,61	1,34	(IPCC, 2006a; Nilsson & Bernesson, 2009)
Nordic											
Winter wheat	6,5	3,72	75	2,79	25	0,93	2,05	1,33	3,45 b	3,38 c	(Bolinder et al., 2007; Kätterer et al., 2011; Nilsson & Bernesson, 2009)
Ley crops, 0 year after oat	1,5	1,88	80	1,50	0	0,00	0,00	1,08	0,00	1,08	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 0 year after wheat	2,5	3,13	80	2,50	0	0,00	0,00	1,81	0,00	1,81	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 1/1 year	9,0	11,25	80	9,00	20	2,25	6,00	6,50	2,25	12,50	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 1/2 year	12,0	15,00	80	12,00	0	0,00	0,00	8,67	0,00	8,67	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Ley crops, 2/2 year	9,0	11,25	80	9,00	20	2,25	6,00	7,43	2,25	13,43	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Oats	4,0	1,98	65	1,29	35	0,69	1,70	1,11	1,98	2,81 d	(Bolinder et al., 2007; Nilsson & Bernesson, 2009)
Winter rapeseed	2,5	2,30	85	1,95	15	0,34	0,58	0,38	2,30	0,96	(Becka et al., 2004; Nilsson & Bernesson, 2009)

[a] *After straw recovery for crop drying: 9.20 for scenarios A,B,C1 and C2; 9.26 for scenario C3.*
[b] *After straw recovery for crop drying: 3.14 for scenarios A,B,C1 and C2; 3.16 for scenario C3.*
[c] *Value for scenarios A, B, C1 and C2. Value for C3 includes extra-root biomass from undersown ley crops: 6.71.*
[d] *Value for scenarios A and B. Value for C1 and C2 includes extra-root biomass from undersown ley crops: 3.19.*