Decentralised Energy Systems Based on Biomass

- a Life Cycle Perspective on Climate Impact and Energy Balance

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Doctoral Thesis
Swedish University of Agricultural Sciences
Uppsala 2015

Acta Universitatis agriculturae Sueciae
2015:15

ISSN 1652-6880
ISBN (print version) 978-91-576-8208-6
ISBN (electronic version) 978-91-576-8209-3

Print: SLU Service/Repro, Uppsala

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Abstract

Replacing fossil fuels with renewable energy sources is recognised as an important measure to mitigate climate change. Residual biomass from agriculture and forestry and short-rotation coppice grown on unused land can be converted to heat, power and fuel without directly compromising production of edible crops. As biomass is mainly produced in rural areas, increased use of biomass-based energy could contribute to job creation and rural economic development.

This thesis investigated whether farmers can generate energy from their own on-farm agricultural and forestry residues for energy self-sufficiency on the farm or for commercial production of district heating or combined heat and power production, with reduced greenhouse gas emissions compared with fossil alternatives. Consequential life cycle assessment methodology was used, with the focus on greenhouse gas emissions and energy balance.

The results showed that arable and dairy organic farms in Sweden can both become self-sufficient in energy by using on-farm biomass residues. Furthermore, decentralised bioenergy systems proved superior to central production based on fossil fuels or large-scale biomass in terms of both greenhouse gas emissions and production costs. The results also revealed large variations (9-97%) in greenhouse gas emissions reduction potential compared with fossil fuels. This variation is partly due to the impact on soil carbon content in soil management systems, where crop residue removal has a negative effect and willow coppice production a positive effect. The input energy requirement for biomass systems is generally higher than for fossil systems, but is typically generated from renewable energy.

Keywords: bioenergy, biomass, decentralised energy systems, crop residues, Salix, biogas, GHG emissions, LCA, organic agriculture

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Dedication

Till mormor Inga-Sara

Nothing in life is to be feared, it is only to be understood. Now is the time to understand more, so that we may fear less.

Marie Curie

Contents

List of Publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I Kimming, M., Sundberg, C., Nordberg, Å., Baky, A., Bernesson, S., Norén, O. & Hansson, P-A. (2011). LCA of energy self-sufficiency systems based on agricultural residues for organic arable farms. *Bioresource Technology* 102, 1424-1432.

II Kimming, M., Sundberg, C., Nordberg, Å., Baky, A., Bernesson, S. & Hansson, P-A. (2014). Replacing fossil energy for organic milk production – Potential biomass sources and GHG emission reductions. *Journal of Cleaner Production* doi.org/10.1016/j.jclepro.2014.03.044

III Kimming, M., Sundberg, C., Nordberg, Å., Baky, A., Bernesson, S., Norén, O. & Hansson, P-A. (2011). Biomass from agriculture in small-scale combined heat and power plants – A comparative Life Cycle Assessment. *Biomass and Bioenergy* 35, 1572-1581.

IV Kimming, M., Sundberg, C., Nordberg, Å., & Hansson, P-A. Vertical integration of local fuel producers into rural DH systems – Climate impact and production costs. Accepted for publication in *Energy Policy*

The contribution of Marie Kimming to the papers included in this thesis was as follows:

I Planned and structured the paper together with the co-authors. Carried out data collection and calculations, and wrote the paper with input from the co-authors.

II Planned, structured and wrote the paper together with the co-authors. Carried out the main part of data collection and calculations. The agricultural production system was dimensioned by co-authors.

III Planned and structured the paper together with the co-authors. Carried out data collection and calculations, and wrote the paper with input from the co-authors.

IV Planned and structured the paper together with the co-authors. Carried out data collection and calculations, and wrote the paper with input from the co-authors.

Abbreviations

AD	Anaerobic Digestion
ALCA	Attributional Life Cycle Assessment
CAP	Common Agricultural Policy
CHP	Combined Heat and Power
CLCA	Consequential Life Cycle Assessment
DH	District Heating
dLUC	direct Land Use Changes
FIT	Feed-in Tariff
FU	Functional Unit
FQD	Fuel Quality Directive
GHG	Greenhouse Gas
GWP	Global Warming Potential
ECM	Energy Corrected Milk
EC	European Commission
ENTSO-E	European Network of Transmission System Operators
EU	European Union
iLUC	indirect Land Use Changes
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
RED	Renewable Energy Directive
RES	Renewable Energy Sources
SRC	Short Rotation Coppice

1 Introduction

1.1 General introduction

According to IPCC, warming of the atmosphere and oceans is now unequivocal. The concentration of greenhouse gases (GHG) in the atmosphere has increased by 40% since pre-industrial times, due to combustion of fossil fuels and emissions due to land use changes, and in the last 40 years, nearly 80% of the increase has been due to fossil fuel combustion and industrial processes (IPCC, 2013).

The European Union (EU) has adopted a target to reduce anthropogenic GHG emissions by 80-95% by 2050, in order to achieve a maximum 2 °C higher average temperature on the planet compared with the baseline of 1990 (EC, 2011). Replacing fossil fuels with renewable energy sources (RES) is one of the most important actions to achieve this, a fact that has been widely recognised by policymakers and public and private investors. Investments in RES are occurring at a rate of about 15% of total global energy investments and have never been higher (IEA, 2014a). According to the European Network for Transmission System Operators (ENTSO-E, 2014), integration of new renewable energy is the main driver of system evolution on the European power market today.

Assessments of biomass potential on a global scale range from about 15% to 50% of the projected global energy demand in 2050 (Popp *et al*., 2014; IEA, 2013; IPCC, 2013). Bioenergy is thus likely to become an important part of the future energy regime.

Compared with wind, solar and wave power, resources that are intermittent (*i.e*. produced and consumed simultaneously), biomass has the benefit of storability, which means that production of heat and/or power can be planned according to consumption patterns. Moreover, biomass comes in many forms (feedstock types) and can be converted to a variety of energy carriers; heat, power or different types of vehicle fuels. These energy carriers can be produced in separate processes or via the biorefinery concept, *i.e.* in integrated processes with multiple outputs.

However, biomass is a bulky material with low energy density (unless dried and comminuted), which limits the feasible transport distance. Therefore, biomass use for internal process energy in the main biomass-producing sectors (forestry and agriculture) can be a way to reduce emissions from those sectors. Forestry and agriculture currently contribute about 24% of global anthropogenic GHG emissions (IPCC, 2013). These mainly originate from enteric fermentation and manure management, nitrous oxide (N_2O) emissions from soil, deforestation and land use changes, but also from the use of diesel and fuel oil.

Organic farms are increasing in number in Sweden and the EU and could have a particular interest in replacing fossil fuels with renewables, as they already have an environmental profile. Taking this even further by achieving energy self-sufficiency would be in line with the organic production principles of utilising local and on-farm resources as far as possible. This could even increase the credibility and competitiveness of organic production as a sustainable alternative.

The climate impact of using biomass for energy purposes has been debated. There is strong scientific evidence in the literature that biomass-based energy systems reduce GHG emissions significantly compared with fossil fuel systems (*e.g.* Ericsson *et al.*, 2014; Fazio & Monti, 2011; Cherubini & Strømman, 2010; Cherubini, 2010; Gnansounou, 2009). However, there are some emissions associated with the production and harvest of feedstock, as well as conversion processes and transportation. Moreover, land use changes due to changes in management, such as crop residue removal, can impair soil quality and hence future yields (Cherubini, 2010; Lal, 2008).

There can also be serious environmental consequences if production of biomass for energy displaces food production on existing land, thereby driving demand for more productive land that can result in destruction of carbon sinks such as forests (deforestation) and peatland. This is referred to as indirect land use change (iLUC).

Deforestation causes a high pulse of carbon dioxide (CO_2) to the atmosphere from the release of carbon built into the trees and the soil organic matter (SOM) content is likely to be negatively affected as carbon becomes more exposed to air via the intensive tillage associated with cash crop production. A consequence of this is a higher rate of mineralisation of carbon to CO_2 and a potential SOM depletion effect, or even soil erosion. Moreover, removing old or pristine forests with high biodiversity poses a threat to rare species of plants and animals and can affect the global hydrological cycle (Ellison *et al*., 2012).

On the other hand, growing short rotation coppice (SRC) such as willow (Salix) on fallow land can be a way to create a carbon sink on arable land (*e.g*. Hammar *et al*., 2014; Ericsson *et al*., 2014). The extensive root system, the addition of leaves and fine roots to the soil and the low-tillage management increase SOM and give a carbon sequestration effect. Carbon sequestration in agricultural soils has in fact been recognised by the IPCC as a climate change mitigation measure in itself (IPCC, 2013).

In order to fully understand the impact of a biomass-based energy system, a thorough environmental assessment is required. Life cycle assessment (LCA) has become an established tool to calculate the life cycle emissions of fuels and energy systems and is described in ISO standards 14040 and 14044. LCA has been widely applied for corporate decisions and as basis for policymaking and policy implementation, such as for calculation of GHG emissions from biofuels under the EU Renewable Energy Directive (RED) (EC, 2009). In LCA, the emissions or resource use associated with a product, process or service in all of its life cycle stages are summarised and normalised by characterisation factors into impact categories, for example climate impact or eutrophication (Tillman, 2000).

In the conventional LCA methodology, attributional LCA (ALCA), the emissions associated directly with a product or service's life cycle stages are calculated in a rather static approach, mostly by applying average emission values (such as the average emissions from the energy mix used in a country or region).

Consequential LCA (CLCA) takes a different, more dynamic approach and has gained ground in recent years. In CLCA, the impact of a *change* to a reference system is assessed (Curran *et al*., 2005; Ekvall & Weidema, 2003; Tillman *et al*., 2000). The multi-market response to the introduction of *e.g.* new energy generation capacity is identified and emissions associated with the increased or decreased marginal production on those markets are assessed.

In this thesis, CLCA was applied to assess the changes in GHG emissions and the energy balance from organic farm production and from heat, power and fuel production when farmers integrated into energy supply chains by using their own biomass residues for energy generation.

1.2 Aim, objectives and structure of the thesis

The main aim of this work was to investigate whether farmers can reduce the climate impact of the agricultural and energy sectors by generating energy from on-farm biomass. The farm-based energy systems studied were aimed either at energy self-sufficiency on the farm, or at commercial local heat and power production systems. The objectives were to assess the global warming potential in a 100-year perspective (GWP_{100}) and the energy balance by applying CLCA for a variety of technical systems based on various biological feedstock types and conversion pathways.

The work comprised two parts, which were further subdivided into four individual studies. The focus of each part is described below. The basic concept of the cyclic material and energy flows in all biomass-based energy systems studied are shown in Figure 1.

In Part 1, the impact of *energy self-sufficiency systems for organic food production* was investigated for an arable farm (Paper 1) and a dairy farm (Paper II). Technical systems in which the farmers could replace all fossil fuel use with the use of crop residues or manure were devised. The production on each farm was assumed to have two main outcomes; commodities in the form of cash crops or energy-corrected milk (ECM), and residues such as straw and green manure. The residues were converted to useful high value energy carriers via conversion processes, and to the extent that the residues contains nutrients (nitrogen and phosphorus), these were returned to the production system (the soil) in the form of digestion residues and ashes. Any by-products or co-products were sold on the market, where they substituted for other products with the same function. The impact on soil carbon and soil emissions was included.

In Part 2, the impact of *the use of farm and forestry residues for local heat and power production* was investigated. Paper III analysed a scenario where a farm supplied a rural village with heat and power in a self-sufficiency system based on agricultural residues (although with the electricity grid as a buffer). Paper IV explored a system where farmers with agricultural and forest production integrate vertically into the supply chain of a local district heating (DH) grid. The consequences on choice of feedstock, conversion pathways and ownership of the DH production and distribution on the GHG balance and heat price were assessed. GHG emissions and costs for fuel supply chains for Salix, straw, forest residues and pellets were calculated.

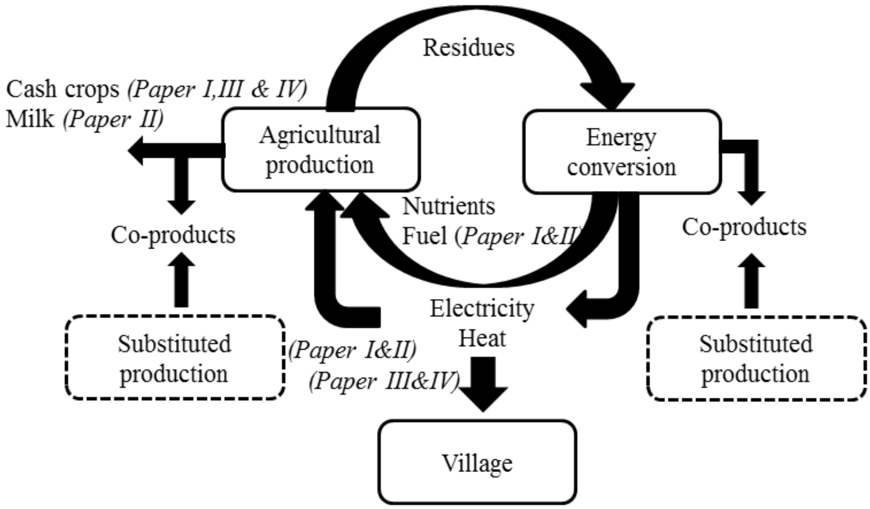

Figure 1. Energy and material flows in the biomass-based systems analysed in Papers I-IV.

The impact categories GWP_{100} and energy balance were included in all studies, as was land requirement. In Paper III, acidification and eutrophication were also included, while a cost analysis was included in Paper IV.

2 Background

2.1 Decentralised energy systems

The concept of decentralised energy supply systems can be defined as “the local supply of electricity and heat which is generated on or near the site where it is used” (Woodman & Baker, 2008). It may of course also refer to on-farm supply of vehicle fuel, as in the systems in Papers I and II. Decentralisation of energy systems can work well for biomass-based heat and power production, as biomass, unless dried and comminuted, cannot be transported long distances (for example to a central power or combined heat and power (CHP) plant) in a cost- or energy-efficient way (Mangoyana & Smith, 2011).

Decentralised energy systems based on biomass may be *e.g.* straw-fuelled heat boilers for residential/commercial/farm use, small-scale district heating or CHP supply systems based on wood chips, small-scale production of biodiesel from the farm’s own oilseed production, or farm-based biogas production to be used as vehicle fuel or for heat and power production in ordinary gas engines.

In 2011, the European Commission (Agriculture and Rural Development group) completed a study on the potential for heat and power production from renewable sources on 800 EU farms (EC, 2011). It concluded that farm-based renewable energy production systems generally have a positive effect on farm income and can increase labour intensity on the farm, *i.e.* create jobs.

Furthermore, although the farm-based renewable energy systems were most profitable in developed and dynamic areas, the highest satisfaction with return on investment was found in less developed areas, indicating that the energy systems were even more welcome there (EC, 2011).

2.2 Bioenergy

2.2.1 Bioenergy use

Sweden has a particularly advantageous situation for use of bioenergy, with extensive forest cover and a well-established forest industry. At present, 33% or 130 TWh of the total final energy use in Sweden is based on biomass (Swedish Energy Agency, 2014). In the DH sector, biomass constitutes more than 40% of the fuel, excluding the organic part of waste (Swedish District Heating Association, 2014).

Most of the biomass used for energy production in CHP or stand-alone heat plants comprises by-products from harvest of roundwood for the timber industry and pulpwood for the pulp & paper industry (Börjesson *et al.*, 2013). Production of biomass for energy is in fact an established industry in itself, and quality aspects and technical and logistical solutions are continuously evolving.

The outtake of forestry residue (tops and branches, and thinning residues) in Sweden is between 7 and 14 TWh/year, still a relatively small proportion of the approximately 155 TWh roundwood harvested annually. The roundwood is mainly delivered to sawmills and the pulp & paper industry, but also for energy production, in the form of low-grade roundwood, and indirectly in the form of by-products, *e.g.* saw dust and bark. The total energy content of the Swedish forest stock is 10 500 TWh and annual growth is 350 TWh. (Börjesson *et al.*, 2013)

However, there is relatively little production of bioenergy from agricultural sources in Sweden. Large-scale, well-established production is limited to the production of ethanol from wheat in Norrköping, which requires wheat from about 100 000 ha per year (Agroetanol, 2014). The wheat is fermented into ethanol, mainly intended for low-blend of ethanol into gasoline.

Biomass use is quite common for heat production in the agricultural sector. From a total of 3300 GWh heat consumed in the agricultural sector in 2012, 600 GWh were produced from roundwood, 345 GWh from wood chips, bark and sawdust, 51 GWh from pellets and briquettes, 66 GWh from grain and 300 GWh from straw. The remainder was produced from oil, gas and electricity, although use of oil has decreased significantly since the previous measurement in 2007 (Swedish Energy Agency, 2014a).

Salix is currently grown on 12 600 hectares (ha) (Statistics Sweden, 2013), producing 180 000 m^3 of fresh Salix wood in 2013. Salix wood chips are mainly used as fuel in heat plants (Swedish Energy Agency, 2014a).

Biogas is produced via anaerobic digestion (AD). In this process, the substrate is converted into a mixture of mainly methane (CH_4) and carbon dioxide by microbes in a heated chamber (Jarvis & Schnürer, 2009). The residue from the AD process is slurry, which contains most of the nutrients from the original substrate and can be used as fertiliser on agricultural soils.

The total Swedish biogas production corresponds to 1.7 TWh, mainly produced at wastewater treatment and co-digestion plants. However, there there is also a small but growing number of farms around the country producing biogas from manure, ley and straw.

About 0.9 TWh biogas is upgraded to vehicle fuel per year in Sweden (Swedish Energy Agency, 2013), which according to current Swedish standards means a minimum methane concentration of 97% (Swedish Standards Institute, 1999). Available upgrading technologies, *i.e.* removing carbon dioxide, include scrubbing, pressure swing adsorption and cryogenic technology, in which methane, carbon dioxide, water vapour and impurities are separated at their different condensing temperatures. The biogas volume is reduced either by compression (CBG) or liquification (LBG) in order to facilitate distribution and use. (Bauer *et al*., 2013)

Bioenergy contributes almost 70% of supplied renewable energy in the EU, mainly in the heat sector, and 8% of total final energy (EUROSTAT, 2012). In 2012, the EU produced 11 million tonnes of wood pellets and consumed more than 15 million tonnes, balanced by imports from (mainly) North America (AEBIOM, 2014).

The EU is also the world's main producer of rapeseed methyl ester, RME (biodiesel) from rapeseed, with more than 40% of global production (Popp *et al*., 2014) and about 8.5% of farmyard manure produced in the EU countries is used for biogas production (Battini *et al*., 2014).

2.2.2 District heating

The DH industry is the largest user of bioenergy in Sweden. Use of fossil fuels dominated in the 1970s but has gradually been replaced by energy from biomass, municipal and industrial waste, waste wood, geothermal heat, heat pumps and industrial waste heat. More than 40% of the fuel used for DH production is currently bioenergy (Swedish District Heating Association, 2014).

A DH system consists of a central heat plant and a distribution grid. Heat can be produced in a stand-alone heat plant, or in a boiler connected to a turbine and generator for CHP production. The boiler consists of a combustion chamber surrounded by a pipe system in which heat is taken up by a medium, often water. The hot water (steam) is led out into a heat distribution system, in low pressure culverts, and heat is exchanged to internal heat distribution systems within buildings (end-users).

In Sweden, the common boiler types are solid biomass boilers adapted to forestry fuels with moisture content between 35-50%. The traditional grate boiler is still common, but among newly built plants the fluidised bed (FB) boilers dominate. These FB boilers use a medium, often sand, that either circulates in the boiler system or creates a bubbling bed in which the fuel is burnt, creating favourable conditions. The FB boilers achieve higher thermal efficiency, but they also require comminuted and homogeneous fuel particles (Strömberg & Svärd, 2012).

The first DH grid was established in Sweden more than 100 years ago, in a hospital area in Vasastan in Stockholm. By the 1950s, the DH concept had been adopted by all major cities in Sweden (Stockholm, Gothenburg, Malmö and others) and the expansion has continued since then. Today, DH in Sweden has about 80% of the market share for apartment buildings, and in total more than 50% of the market share for heat (Swedish District Heating Association, 2014).

Before the market deregulation in 1996, all DH utilities in Sweden were owned by municipal authorities. Today there are over 220 energy utilities that produce and distribute heat, both private and public, although most are in fact still municipal. EON and Fortum are the largest private actors, and together with Vattenfall AB and Göteborgs Energi are the largest overall suppliers, with several grids across the country (Fortum has concentrated ownership to the Stockholm area). (Swedish Energy Market Inspectorate, 2014)

At the other end of the scale, there are DH systems with 10-20 GWh heat delivered per year, such as Lekeberg Bioenergi AB near Örebro, which is owned by local farmers who supply most of the fuel from their own farms and forests.

The DH sector is currently facing challenges such as saturated markets, energy efficiency measures (reducing demand), competition from individual heat solutions such as heat pumps and low consumer trust due to lack of competition on the market. Distribution grids for heat were long considered to be natural monopolies, as multiple grids in a region would be too expensive to benefit consumers. Hence, each owner of a DH grid had a monopoly. However, the monopoly situation in the DH sector has led to weak incentives to keep costs and prices low. (Löfblad *et al*., 2013)

Third party access (TPA) is a system that obliges the owners of distribution grids (for heat, gas, electricity *etc*.) to allow regulated access to third parties for energy distribution to end-users. A TPA system for the Swedish DH market has been discussed and investigated for several years, and came into force on 1 August 2014, as an amendment to the Swedish Act on District Heating (Swedish Code of Statues, 2014).

The DH concept can work in large cities and in small-scale, decentralised settings, and can provide an opportunity in rural areas for commercial production of biomass-based heat.

In the EU as a whole, only 13% of heat is supplied by DH, mainly in Denmark, Norway, Sweden, Finland and Poland. Most heat for the residential/commercial sector, over 40%, is currently produced by natural gas in on-site boilers (Connolly *et al*., 2013).

2.2.3 Bioenergy policy

The EU's energy policy is defined by the 20-20-20 goals adopted in 2009; 20% reduction in GHG emissions, 20% energy from renewable sources and 20% energy efficiency by 2020 (EC, 2009). In October 2014, EU member states agreed a new, more ambitious target of 40% GHG emissions reductions, 27% renewable energy and 27% energy efficiency (EC, 2014).

The main instrument for reaching the target on GHG emissions reductions is the cap-and-trade system, the EU Emissions Trading Scheme (ETS), effective from 2008. The ETS aims at creating a market-based and cost-efficient system for emissions reductions by requiring industries (with certain minimum production capacity) to obtain an emissions certificate for all their CO_2 emissions. Some emission rights are allocated to the industries at the start of a trading period, the rest are traded on the market. High emitters are punished by having to buy the emission certificates, which in effect adds a price to release of carbon, and low emitters can sell emission certificates, which in effect can add to the profitability of renewable energy investments. However, the ETS system has suffered from an oversupply of emissions rights, with plummeting prices as a result, hardly achieving the intended steering effect (EC, 2012).

The Renewable Energy Directive, RED (Directive 2009/30/EC) aims at promoting biofuels for transport. Certain sustainability criteria must be fulfilled in order for a fuel to be denominated "biofuel"; a minimum of 35% emissions reductions for liquid and gaseous biofuels must be reached compared with the fossil alternative, with this requirement increasing to 50% by 2017 and 60% by

2018. The RED also describes methodology for calculating the GHG emissions, based on LCA (EC, 2009a).

There is currently a cap of maximum 7% first-generation biofuels (from primary crops such as wheat or rapeseed), due to concerns about iLUC issues arising from replacement of food crops. A non-binding goal of 0.5% advanced biofuels from lignocellulosic material and agricultural residues has been set. The EU Fuel Quality Directive, FQD (Directive 2009/30/EC) allows 10% blend of bioethanol into petrol and 7% biodiesel into conventional diesel. The FQD also stipulates that fuel suppliers must reduce their GHG emissions per energy unit by 6% by 2020. (EC, 2009b)

The EU Common Agricultural Policy (CAP) does not support production of energy crops directly, since the energy crop premium and the set-aside scheme (in which energy crops could be grown on set-aside land) was abolished in 2009. It was replaced by the single area payment, which means that crop production obtains support regardless of purpose (food, energy, fibre *etc.*).

The investment programme NER300 is a funding programme for large-scale investments in low-carbon technologies. There are also national support systems partly funded by the EU, such as biogas investment support in Sweden through the rural development programme 2007-2013 (Swedish Board of Agriculture, 2009).

Besides EU policy, member states have their own ways of subsidising renewable fuels. The feed-in tariff (FIT) in Germany is a successful, but expensive, example of energy policy. Introduced in 2000, the system guaranteed a certain selling price over 20 years for electricity from renewable energy sources. It had a substantial impact on the market, with primary energy production from renewable sources more than tripling between 2002 and 2012. This made Germany the largest producer of renewable energy in the EU, comprising more than 18.6% of the total renewable energy produced (Sweden is third, with approximately 10.4% of total renewable energy production; Eurostat, 2014). Thousands of biogas installations were installed in Germany as a consequence of FIT schemes, but also in Italy, France and the UK (EBA, 2013). In 2012, there were in total 13 800 biogas plants in the EU (EBA, 2013).

Sweden has chosen a different approach when it comes to energy policy. In 1995, a CO_2 tax was imposed on use of fossil fuels, from which renewable fuels are exempt. Renewable fuels are also exempt from energy tax. In 2003, the green certificate system was implemented. This quota system aims at increasing the share of renewable energy sources by 25 TWh by 2020.

Electricity producers are obliged to use a set quota of renewable fuels or otherwise purchase certificates from producers with a higher share of renewable production than required. Since 2012, Norway is also included in the green certificate system.

After the introduction of the quota system, many stand-alone heat plants were converted to CHP (Swedish Energy Agency, 2014b). The tax system and quota systems have generally been considered successful in creating incentives for use of renewable fuels in Sweden (McCormick & Kåberger, 2007).

2.3 Agriculture

2.3.1 Production, structure and policy

In 2012, the 28 EU member states produced cereals on 57 million ha, oilseeds on 11 million ha and, in addition, potatoes, sugar, rice, fruit and vegetables (EU DG-ARD, 2012). Sweden has in total 2.6 million ha of agricultural land, of which about 1 million ha are used for cereals, 1 million ha for ley, 160 000 ha for fallow and the rest for production of oilseeds, potato, sugar beet *etc.* (Statistics Sweden, 2014).

Swedish production is currently distributed over almost 64 000 agricultural holdings, with average area 39 ha. This number has decreased since 2007, when there were over 72 000 holdings (Swedish Energy Agency, 2014a). However, 39 ha farm area is still large compared with the EU as a whole, where the average farm size is 12 ha (EU DG-ARD, 2012). The average farm size varies widely between the EU member states.

All agricultural policy in the EU is gathered under the CAP and little is left to the individual member state (EC, 2014). The CAP consists of three pillars: market support, income support and rural development. The purpose of the market support mechanism is to ensure that there is a market for European agricultural commodities. The income support comprises direct payments to farmers and, since the CAP reform in 2003, is decoupled from production. It is now given in the form of single farm payments, coupled to cross-compliance with certain requirements on animal welfare and environmental/health standards.

Under the national rural development programmes, farmers can get support for different investments, for example farm-based biogas production. Each member state co-designs and co-funds these development programmes, whereas market and income support are funded by the EU.

The agricultural sector has undergone sweeping structural change in the past few decades. In Sweden, the area used for crop production decreased by about 10% after the introduction of the EU single farm payment and the number of holdings with livestock has also steadily decreased (Statistics Sweden, 2014). Farms are also generally becoming larger, fewer in number, more specialised and with lower labour intensity. More than half of the agricultural holdings in Sweden require less than 800 hours of labour per year (Statistics Sweden, 2014).

Agriculture is also shrinking as a share of gross national product in many member states, and as a share of the EU budget. The CAP currently takes up 40% of the EU budget, which represents a major decrease from 75% three decades ago (EC, 2014).

2.3.2 Organic farming

In the EU, the area cultivated according to organic principles has grown rapidly in the past decade, along with rising demand for organic food, but still only makes up 5.4% of total agricultural land (European Network for Rural Development, 2014). In Sweden, 15.7% of arable land is managed according to organic principles and about 8% of dairy cows are organically managed. Globally, about 1% of arable land is certified according to some organic label (Willer & Kilcher, 2012).

Regulations for organic agriculture in the EU are set by Council Regulation No. 834/2007 (EC, 2007), in which the definition of organic farming is "an overall system of farm management and food production that combines best environmental practices, a high level of biodiversity, the preservation of natural resources, the application of high animal welfare standards and a production method in line with the preference of certain consumers for products using natural substances and processes" (EC, 2007).

Important principles are elimination of the use of mineral fertilisers and chemical pesticides. Nutrients can be added *e.g.* by including a nitrogen-fixing ley in the crop rotation, or by using an organically approved fertiliser, for example manure. Research has also shown that human urine, mussels or other nitrogen-rich biological products can be added to agricultural fields as fertiliser, however the spreading of urine is not permitted (Spångberg, 2014). The organic principles also include minimised resource use and increased use of renewable resources, but so far there is no specific requirement on the use of renewable energy. (EC, 2007)

In Sweden, the established organic certification label is KRAV. Since 2014, KRAV has set special requirements related to energy use for organic farms that use more than 500 GWh per year or have more than 100 cattle; such farms must map their energy use, strive for energy efficiency and develop key performance indicators. Furthermore, 100% of electricity must be from renewable energy sources, either their own production or certified sources (KRAV, 2014).

There are studies that indicate that organic farms require less energy than conventional farms (Tuomisto *et al*., 2012). This is partly due to the exclusion of mineral fertilisers, which are energy-intensive to produce. According to an assessment of conventional agricultural systems in 15 European countries, about 47% of the total energy input for crop production is used for production of fertilisers, of which almost 80% for nitrogen fertiliser production (Alluvione *et al*., 2011).

Organic farms may require more mechanical weeding, on the other hand, and produce less output per hectare. Hence, if calculated per unit of output, organic production may be more energy-intensive that conventional production. However, when comparing an organic and a conventional farm of the same area, the conventional farm might be the largest energy user (Tuomisto *et al*., 2012).

2.4 Assessing bioenergy systems

2.4.1 Assessment methods

Several methods for environmental assessment have been developed, such as Strategic Environmental Assessment (SEA), Environmental Impact Assessment (EIA), Material Flow Analysis (MFA), the Public Goods (PG) tool and the Carbon Footprint (CF) (Leach *et al*., 2012; Jensen, 2012; Finnveden *et al*., 2009). The scope of these tools differs, *e.g.* the PG tool is used to visualise the impact of milk production, based on real farm data, by 11 "spurs" in a radar diagram. These are soil management, biodiversity, landscape & heritage, water management, nutrient management, energy & carbon, food security, agricultural systems diversity, social capital, farm business resilience, and animal health & safety (Leach *et al*., 2012). In contrast, the CF focuses only on the carbon balance of a product or service (Jensen, 2012).

However, the most established methodology today for both corporate purposes and policy making is LCA. LCA thoroughly assesses the impacts of products, processes or services in predefined impact categories such as climate impact, acidification, eutrophication and resource use, and is hence an extremely comprehensive method when strictly applied. Among other uses LCA is prescribed in the RED for calculation of GHG emissions from biofuels.

The LCA methodology has been standardised in ISO 14044 (CEN, 2006), and guidelines have been developed by *e.g.* the UNEP/SETAC Life Cycle Initiative.

2.4.2 LCA overview

An LCA process has four basic stages: 1) Goal and scope definition, 2) life cycle inventory (LCI), 3) life cycle impact assessment (LCIA) and 4) interpretation of results (Tillman, 2000). Stage (2) includes collection of emissions data and stage (3) includes categorisation into impact categories (such as GWP, acidification potential, land use *etc*.), and normalisation to a common unit (such as CO_2-equivalents (CO_2-eq.) or ha). This is done by characterisation factors that depend on the potency of the emissions to create damage – for example, GWP_{100} measures the potential to affect the energy balance of the planet via changes in radiative forcing from the existence of GHG (such as CO_2, CH_4, N_2O and more) in the atmosphere over 100 years.

There are two main orientations of LCA; ALCA and CLCA (*e.g.* Finnveden *et al.*, 2009; Curran *et al.*, 2005; Ekvall & Weidema, 2003). ALCA is the traditional approach and involves attributing emissions to each process step as they occur – from transportation, conversion processes and use. Average values for input factors apply, such as the energy mix used in the relevant geographical region. The environmental burden is allocated between main products and co-/by-products based on physical or economic (or other) properties. The ALCA approach is often used for example for labels and standards, and for calculation of GHG emissions under the RED.

In CLCA, the change to a reference system when a new product or service is introduced is assessed. For CLCA, it is important to understand the multi-market response to the introduction of the new system, *e.g.* the type of electricity production responsible for the long-term increase/decrease as a consequence of the newly built (perhaps more environmental/climate friendly) electricity production capacity. When output from existing production capacity is reduced due to new investments, the reduction is referred to as substituted production, and the potential reduction in emissions is ascribed to the new system. Allocation is not applied in CLCA.

CLCA can be used both as a basis for policymaking and for monitoring the effectiveness of a policy. For example, Bento & Klotz (2014) showed by CLCA that different biofuel policies in the USA resulted in emissions ranging from -16.1 to 24 g CO_2-eq./MJ maize ethanol entering the market as a result of the policy.

2.4.3 Impact categories and characterisation factors

A certain set of emissions can lead to the same environmental impact – for example, greenhouse gases, GHG (*e.g.* CO_2, CH_4, N_2O) contribute to warming of the planet by "trapping" heat from solar irradiation inside the atmosphere. The emission metric used for GHG is Global warming potential (GWP). The GWP metric measures radiative forcing; *i.e.*, the difference between the sunlight absorbed by the planet and the sunlight radiated back to space, integrated over a defined time horizon. Positive radiative forcing leads to near-surface warming of the planet, negative forcing leads to cooling. The emitted gases are assigned characterisation factors determined by how strongly they contribute to this greenhouse effect in a given time span, compared to CO_2, and given in the unit CO_2-equivalents.

The time span is often set to 100 years, GWP_{100}. For example, 1 kg of CH_4 emissions that remains in the atmosphere for 100 years corresponds to 23 kg CO_2 in the atmosphere for 100 years, and 1 kg of N_2O emissions has the potency of 298 kg CO_2 in the atmosphere (IPCC, 2006). The GWP_{100} metric was adopted by UNFCCC in the Kyoto Protocol, and became a default metric for LCA practicioners. (IPCC, 2014)

2.4.4 Functional units

Functional unit, FU, is the unit to which emissions or resource use are related, *e.g*. emissions per MJ or resource use per hectare. The choice of FU is important, as it affects the absolute results, the applicability of the results and the comparability to other studies. It is important to be consistent in the use of FU for a study in order to compare the same function of a service or product, but also to adapt it to the aim of the study.

FU for bioenergy can be categorised into four different types; output-related, area-related, input-related and on a yearly basis (Cherubini & Strømman, 2010). For example, when comparing biofuels to biodiesel in a CLCA, the FU per vehicle-km could be considered appropriate because it includes the actual function, *i.e*. driving. On the other hand, when comparing the use of biomass for energy with production of food and fibre, the final energy production per hectare may be most relevant because land is a constrained resource.

In order to make the study independent of feedstock, the appropriate FU could be output-related, such as emissions per MJ or vehicle-km An FU per unit, such as kg or MJ, could be used in order to make the result independent of conversion processes (Cherubini, 2009).

2.4.5 System boundaries

A careful definition of system boundaries is critical in LCA studies (*e.g.* Finnveden *et al*, 2009; Schlamadinger *et al*, 1997). All processes involved or affected in the production of the biofuel should be accounted for, and multiple processes need to be handled by allocation or substitution. Allocation is used in ALCA but not in CLCA, where instead the market response to placing by-products and co-products on the market and the consequence on the environmental impact are quantified. This impact is often beneficial, since other emissions can be avoided by substituting production. The impact of substitution is accounted for by a negative value if emissions are avoided or a positive value if emissions are added. The negative value could be *e.g.* avoided fossil fuel use.

2.5 Bioenergy LCA studies

There is an extensive body of LCA studies of bioenergy in the literature, not least on the topic of climate impact. The results of these studies show very large variations, even for similar bioenergy chains. These differences can be attributed to methodological choices, input data used, reference systems or simply the quality of the work (Muench & Guenther, 2013; Cherubini, 2009).

A review conducted in 2010 showed that the majority of bioenergy LCAs published at that time focused on climate impact and energy balance, whereas only 9% considered the land use category, *i.e.* iLUC (Cherubini & Strømman, 2010). About 25% take direct land use change (dLUC) into account. Consequential LCA is the most common approach for LCA of bioenergy, used in 75% studies.

There are in fact more studies conducted on transportation fuels based on biomass rather than on heat and power from biomass, even though the former only makes up about 5% of total bioenergy use and 3-4% of total transport fuel consumption (Popp *et al.*, 2014). There is a particularly high frequency of studies on bioethanol and biodiesel production, and on fuels from lignocellulosic substrates (Popp *et al.*, 2014).

2.5.1 GHG emissions

The volume of GHG emissions produced in biomass-based energy generation systems depends on choice of feedstock, conversion technology and plant efficiency, distribution chains, by-product generation, how much the main or co-products can substitute other production, and how emission-intense the substituted production is.

Power production based on coal or natural gas is often used as a reference in LCA studies (mainly in CLCA as they more often constitute marginal production than the average production mix). Coal condensing plants emit about 230 g CO_2-eq./MJ (assuming 48% electric efficiency) and natural gas-based production 120 g CO_2-eq./MJ (assuming 58% electric efficiency) (Giuntoli *et al*., 2014).

In comparison, a review paper by Varun *et al*. (2009) found that emissions from electricity generated by biomass in large-scale condensing power plants can range between 10 and 49 g CO_2-eq./MJ electricity produced (includes also the biomass part of low-blend into coal). In a more recent review paper, Muench and Guenther (2013) found a variation of 0.3 to 193 g CO_2-eq./MJ electricity produced, with a median of 38.19 for electricity production from biomass. The high end of the scale is due to the impact of iLUC such as deforestation.

Emissions from stand-alone heat production vary less, according to Muench and Guenther (2013), from 1.65 to 21.39 g CO_2-eq./MJ heat (median 8.2 g CO_2-eq./MJ). The median for CHP production is -25.18 g CO_2-eq./MJ heat and the range is -47.5 to 8.6 g CO_2-eq./MJ heat. The emissions value can become negative when power production from biomass CHP substitutes power produced from fossil fuels.

According to a review in 2009, biofuels in the transportation sector produce GHG emissions within the range 15-195 g CO_2-eq./km. The lowest emitters are biodiesel based on Fischer-Tropsch technology, bioethanol from lignocellulosic crops (sometimes referred as second generation biofuels), and biogas (Cherubini, 2009). The highest emitters are when severe iLUC effects can be expected, for example deforestation. For comparison, diesel cars emit 82 g CO_2-eq./MJ (Gode *et al*, 2011), which corresponds to 145 g CO_2-eq./km assuming diesel consumption of 0.5 l/km and diesel energy content of 35 MJ/l.

Biogas production can have multiple benefits for the climate; it can substitute fossil fuel use, reduce methane emissions from manure management systems at farms, and substitute production of artificial fertilisers. Lantz & Börjesson (2014) calculated the GHG emissions from biogas produced from agricultural residues and upgraded to vehicle fuel quality, *i.e*. 97% methane and compressed (CBG) or liquefied (LBG), to 8.2 g CO_2-eq./MJ when including the substitution effects. Direct emissions from the biogas system (excluding the fertiliser substitution effect, but including reduced emissions from manure management) were 17.1 g CO_2-eq./MJ. Börjesson & Berglund

(2006) pointed at the fact that life cycle emissions from biogas for vehicle fuel use depends on the substrate type (manure, straw, ley, sewage sludge *etc*.), the conversion efficiency of the system, the methane slip in the upgrading process *etc*., and hence results from one analysis of an energy system may not apply to another. Similarly, Huttunen *et al.* (2014) concluded in a review of biogas LCA studies that substrate type, transport distance, end use of the gas, utilisation of digestion residues and system boundaries strongly influence LCA results on climate impact.

Forestry residue supply chains for combustion in heat/CHP plants are attributed with emissions in the production phase, *i.e*. the difference in emissions compared with a system where residues are left on the forest floor. Emissions comprise forwarding, transport, storage, conversion and soil carbon changes in the forest soil. Lindholm *et al*. (2010) estimated the GHG emissions from harvest of logging residues (loose or in bundles) in southern Sweden to be 7.6-8.5 g CO_2-eq./MJ, including soil carbon effects, which had a large impact on the results (similar calculations, without including soil organic carbon effects, gave emissions of 2.3 g CO_2-eq./MJ for central Sweden; Berg & Lindholm, 2005). Berg *et al*. (2014) found a range emissions between 7-19 kg CO_2-eq./m^3u.b. (ca 1–2.8 g CO_2-eq./MJ) for forest supply chains in Germany and Sweden, depending on the topography of the harvested area and mechanisation of the harvest operations. Energy use varied between 83-206 MJ/m^3u.b, of which 35-60% was from transportation. The highest values were for Sweden. This corresponds to about 1-3% of the energy contained in the wood.

Use of agricultural residues, in particular straw, for energy generation may cause less or no emissions in the production phase compared with use of primary agricultural products, but crop residue removal from fields has an impact on soil organic carbon (SOC) content and nutrient concentrations. This can affect the future productivity of the soil (Cherubini, 2009). Many LCA studies unfortunately omit this aspect, but Fazio and Monti (2011) found that removal of crop residues for biofuel production increases emissions from land use by 20%, mainly due to the increased use of fertilisers. According to Gabrielle *et al.* (2014), the nutrient requirements of lignocellulosic crops are still poorly known, which adds to the uncertainty.

Cherubini and Ugliati (2010) expressed the effect of crop residue removal as the sum of changes to SOC, the emissions from additional production required elsewhere due to lower productivity, the use of fertiliser to replace the nutrients removed with the crop residues, and the reduction in nitrous oxide emissions from crop residues that would have been ploughed back into the soil if they had not been used for biofuel production. The actual impact of crop residue removal depends strongly on factors such as soil type, soil climate, time scale and management practices (Cherubini, 2009).

2.5.2 Energy balance

Similarly to GHG emissions, the cumulative energy requirement from biofuel use depends on feedstock choice, conversion technology and efficiency, fuel quality and engine efficiency. The latter only becomes visible in studies that take into account the use phase (well to wheel) with a functional unit such as energy use per 1 km of driving (as opposed to *e.g.* per litre or MJ of fuel produced). In general, bioenergy systems require higher energy input than fossil fuel systems on a life cycle basis, due to the field operations and transportation required during the production phase.

Cherubini (2009) reported a range of cumulative energy requirement of 3.5-13 MJ/km in LCA studies on various types of biofuels, which can be compared with the range of 1.3-2.4 MJ/km reported for petrol and diesel. However, while the input for petrol and diesel is almost exclusively of fossil origin, the opposite is true for biofuels, which often rely more on renewable energy input, *e.g.* use of biomass for grain drying or biogas chambers heated with a stream of the produced gas.

There are LCA studies of biofuels that indicate that in some biofuel supply chains, the fossil input exceeds the energy output in the final energy carrier, for example ethanol made from maize, switchgrass or wood biomass in the US and soy bean/sunflower biodiesel (Cherubini & Stromman, 2010). Other studies contradict this, *e.g.* Blottniz and Curran (2007) found that ethanol from maize stovers in the USA indeed requires less energy to produce and distribute than is delivered by the fuel. Production technologies, assumed conversion efficiency, and site-specific conditions for each supply chain studied can play a large role in such contradictory results, but also methodological choices, *e.g.* assumptions on substituted production, allocation and system boundaries.

2.5.3 Land use and iLUC

The land use category indicates how much land area is occupied by production of the biofuel product under study, and is closely linked to energy yield per hectare. There is typically higher energy output per unit area in tropical regions than in temperate regions, since the crop productivity is higher (Blottnitz & Curran, 2007). This may mean that biofuel from such regions can appear favourable if the functional unit is area-related, *e.g.* km driven on biofuels produced on 1 ha. For example, a Salix plantation yields 198 GJ/ha assuming

10 tonnes of dry matter (DM) per ha and year (Gonzáles-Garcia *et al*., 2012), but the yield ranges between 5-11 tonnes DM/ha and year depending on growing conditions (Gabrielle *et al*., 2014). Since doubling yield does not mean doubling the necessary field operations, the energy output per ha can range between approximately 100-200 GJ/ha and year. From sugarcane, about 120 GJ of ethanol can be produced per ha and year, while wheat and oilseeds produce about 42 GJ/ha and year, corresponding to yield of 12.3, 4.2 and 2.7 tonnes DM/ha and year, respectively (Gabrielle *et al.*, 2014).

Land use mostly refers to use of arable land that could have been used for food production. Displacement of food production can start a complex chain of displacement mechanisms across the world, since the market for agricultural goods is largely global and food demand can be considered inelastic (Kløverpris, 2008). The chain of displacement can end in *e.g.* deforestation, because more arable land eventually is needed (the iLUC effect).

Reinhard and Zah (2011) showed that biodiesel from rapeseed in Switzerland can cause emissions within the range -175 to 329 g CO_2-eq./MJ fuel. Substitution of high-emitting products can result in a negative emissions balance, whereas deforestation or other iLUC effects cause the higher emissions. By solely taking into account the emissions associated with the production phase, Reinhard and Zah (2011) also estimated that biodiesel caused 62 g CO_2-eq./MJ, which can be compared to ca 85 g CO_2-eq./MJ for petrol and diesel (Gode *et al*, 2011).

The land *requirement*, as opposed to land *use*, measures the area from which residues from food crop production are harvested, resulting mainly in dLUC, such as an impact on soil carbon dynamics, but not in displacement of cash crops. However, dLUC can potentially lead to iLUC if the biomass removal results in lower productivity rates, driving a demand for new crop producing land.

Globally, about 2.5% of arable land, or 40 Mha, is used for primary bioenergy production, with implications both for direct and indirect land use change. In Europe, the corresponding figure is about 3 Mha, mainly for biodiesel and bioethanol, *i.e*. rapeseed and wheat (Popp *et al*., 2014). About 0.1 Mha is dedicated to SRC production.

3 Methodological approach

This chapter describes more specifically how CLCA was applied and used in Papers I-IV. In particular, the emissions calculation methodologies, soil carbon model and some of the most important assumptions used in the papers are described.

3.1 LCA approach

CLCA is generally considered to be more suitable for assessment of new systems or for policy-making, as it measures the impact of a change, whereas ALCA is considered more suitable for environmental labels and certification, as it may be easier to communicate and seem more transparent (*e.g.* Soimakallio *et al.*, 2011; Finnveden *et al.*, 2009). Since Papers I-IV assessed the change from an existing energy system to a biomass-based energy supply system, CLCA was consistently applied.

3.2 Functional unit

In Papers I and III, the FU was total emissions on a yearly basis from the entire farm system. The main reason for this was to capture the entire agricultural system without having to apply too many substitution effects or cut-offs. In the cash crop systems, there are several farm outputs and it is not a straightforward matter to determine which one that is the main product.

In Paper II, on the other hand, the FU was on an output basis, 1 kg energy-corrected milk (ECM), because milk is clearly the main output from the farm, and the purpose was in fact to investigate the climate impact from its production. Similarly, in Paper IV the FU was emissions on an output basis (1 GJ heat), because it was desired to investigate emissions and costs of the heat delivered to an end-user. The output basis also facilitates comparison with other studies in the literature.

The impact categories considered were GWP_{100}, and energy balance. Land requirement has been calculated in all studies, and in Paper III also acidification and eutrophication. The characterisation factors applied are based on the IPCC Fourth Assessment Report (IPCC 2006) (Table 1).

Table 1. *Global warming potential in a 100-year perspective (GWP_{100}) of the major greenhouse gases carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)*

	CO_2	CH_4	N_2O
GWP_{100} (CO_2-equivalents)	1	23	298

3.3 Emissions from arable land

3.3.1 Methane and nitrous oxide

Calculating emissions of nitrous oxide and methane from the soil and any fertilisers applied to the soil is not a straight-forward task due to lack of verified models and data with high geographical resolution. The most common methodology today for calculation of emissions from managed soils is that developed by the IPCC, in which it is assumed that a certain fraction of applied fertiliser will volatilise directly, or leach and volatilise at a later stage.

Methane emissions during storage and spreading of manure and digestion residues were also calculated using the IPCC methodology. Nitrous oxide and methane emissions from manure and digestion residues were calculated by the following equations:

$$N_2O_{emitted} = D_{ts}\ x\ Frac_{N-tot}\ x\ \left(\frac{44}{28}\right) x\ EF$$

$$\mathrm{CH}_{4\ emitted} = D_{ts} x\ B_0 x\ 0.71 xMF$$

where D is the volume of digestion residue or fresh manure, $Frac_{(N\text{-}tot)}$ is the fraction of nitrogen available, 44/28 is the conversion factor for nitrogen to nitrous oxide, EF is the emissions factor, B_0 is maximum methane production capacity, 0.71 is the density of biogas and MCF is the methane conversion factor. (IPCC, 2006)

3.3.2 Soil carbon dynamics

An adapted model for calculating soil emissions from crop rotations was developed for this work. The methodology is based on the ICBM model

(Andrén & Kätterer, 2001). Using the ICBM model, the carbon dynamics were simulated during three crop rotations (21 years), given an assumed initial carbon content in year 1 and assumptions on soil type, soil climate and tillage. In each year and for each field, the carbon content will increase or decrease based on the parameters mentioned above, in combination with the amount of carbon input to the soil in the form of crop residues and fertilisers. The resulting carbon content after each year's cultivation represents the input for the next year. The simulation was also applied to the Salix plantation in Paper I, which was assumed to have a lifetime of 21 years. The parameters were used in a set of equations validated in long-term field trials in Sweden.

The most important parameters in the model are carbon input (from crop residues such as straw, rhizodeposition and manure), humification coefficient and soil climate. The humification coefficient, or the fraction of material converted to resistant soil organic matter, depends on the ratio of the carbon sources, and is lower for crop residues than for manure (Andrén & Kätterer, 2004). The mineralisation rate of carbon increases with increasing frequency of soil tillage, and is consequently higher for annual crops in the crop rotation than for perennial crops. This is accounted for in the model through the soil climate parameter, which is based on climate data for the region (air temperature, precipitation and potential evapotranspiration).

3.4 Land requirement

Arable land is a constrained resource, and extensive production of agricultural biomass for energy purposes could displace food production. Given a relatively inelastic food demand and a growing population on the planet, this means that new arable land will eventually have to be taken into production. This can be done by *e.g.* deforestation, which has serious environmental impacts. However, iLUC may not be an issue in regions where there is a 'land buffer'.

In 2008, the Swedish Board of Agriculture quantified the amount of Swedish arable land or pasture land currently not in use and found it to be 500 000 ha, while overproduction of ley crops occupied 200 000-300 000 ha (Johnsson, 2008). The area of fallow land in 2013 was at least 160 000 ha, according to Statistics Sweden (2014).

Land for production of SRC can hence be available without directly compromising food production in Sweden. In this thesis, a basic assumption was that land is available for SRC production without the implications of iLUC. Land requirement was however calculated, for two main reasons: to determine the area required for a system (and thereby assess the practical feasibility of the system) and to determine the area affected by residue removal and susceptible to soil quality changes.

3.5 Machinery

Emissions from tractor use, crop drying and transport in the reference scenarios were calculated by applying emission factors for the fossil fuels (found in the literature) used in the different types of machinery, multiplied by the actual consumption for each activity. Use of the biomass-based energy carriers in the respective machines or technical systems was assumed to be a carbon-neutral process (as is common practice in LCA). However, in Paper I formation of methane and nitrous oxide in engines was included.

3.6 Process energy and marginal electricity production

In Part 1 of the thesis, the energy for conversion of biomass to energy in each scenario was generated by the system itself, creating a circular (self-sufficiency) system. A self-sufficiency system for the farm only was thus created (although with an electricity exchange to the national grid). In Part 2, a local energy supply system with system boundaries drawn around the farm and the village (also with the power grid) was created.

In the fossil fuel-based reference systems, the emissions factors per unit of energy included upstream activities, such as extraction, refinement and distribution.

Energy use is a critical parameter in LCA, since most processes require the input of energy. Processes may also entail an output of energy, as in this case.

In CLCA the question is which type of electricity production is either substituted/decommissioned or increased/commissioned by the new system. This is referred to as the marginal electricity production. Marginal production can simply be categorised into two types, the built or long-term margin, which refers to new production capacity, and the short-term margin, which is based on daily fluctuations in demand.

The latter is quite straight-forward to determine based on merit order in the European energy system. However, it is the built margin that is of interest in LCA when analysing systems with a lifetime of several years/decades.

In 2006, an assessment of the European energy system showed that power production capacity based on natural gas dominated in planned investments. Coal-based production dominated amongst plans that were to be decommissioned within coming decades (Kjaerstad & Johnsson, 2007).

Natural gas, with emissions of 120 g CO2-eq./MJ, was therefore used as marginal production capacity in Papers I-IV.

3.7 Scenario building

The scenarios developed for each study were based on energy technologies that were either commercially available at the time of the study, or deemed to have potential to become commercially available within the next five years. The focus was on building realistic scenarios from a technical and logistical perspective, but these were not constrained by their economic viability in today's market and policy situation.

The setting, data and assumptions used apply to south-western Sweden, but in most cases the general conclusions could also be valid for other areas and regions. Heat and power production systems with natural gas as fuel were chosen as reference systems, in order to make the systems relevant to the EU as a whole; natural gas is the most common fuel for heat production in the residential/commercial sector in EU, currently making up 44% (Connolly *et al.*, 2013).

The farms in Part 1 were dimensioned based on a specific typical size (arable) and typical number of cattle (dairy) in order to be representative. The crop rotation was developed in cooperation with the Swedish Institute for Agricultural and Environmental Engineering (JTI). The number of houses in Part 2 was chosen on a somewhat more arbitrary basis, as the size of rural villages can differ from few houses to several hundred houses.

4 Part 1: Energy self-sufficiency

4.1 Basic system description

The organic farms studied in Part 1 were assumed to be located in the County of Västra Götaland with respect to weather data, soil quality, crop yield, typical farm size and production. The farms were assumed to apply organic production methods according to the criteria stipulated in Council Regulation (EC) No. 834/2007. Paper I studied an arable farm, and the FU was the entire farm output for one year. Paper II studied a dairy farm, and the FU was 1 kg ECM at the farm gate.

The crop rotation applied on the arable farm is shown in Table 2. The cultivated area on the farm was 200 ha. As a substitute for artificial fertilisers (not permitted in organic farming), the crop rotation included a nitrogen-fixing ley crop twice in the seven-year rotation, a practice also referred to as green manuring. In the reference scenarios, the green manure crop was ploughed back into the soil, while in two of the alternative energy scenarios it was harvested and used as substrate for biogas production.

Table 2. *Crop rotation and average yield on the arable farm studied in Paper I (Hansson* et al.*, 2007)*

Yield (kg dry matter/ha yr)[a]	Field beans	Oats	Ley [b]	Rapeseed	Winter wheat	Ley [b]	Rye
Crop yield	2400	3200	6000	2000	3500	6000	3200
Straw yield	-	2756	-	1818	2846	-	4571
Total ploughed down biomass	3600	4800	1500	6000	7000	1500	4800

[a]Crops dried to 86% dry matter, except ley which is in kg dry matter
[b]Green manure

The dairy farm was assumed to have a herd of 100 cows, with 25% recruitment rate. In accordance with organic principles, it was assumed to be self-sufficient in organically produced forage, of which 50% must consist of silage and grazing according to KRAV requirements. The two crop rotations tested are shown in Table 3. Outputs from the farm in addition to milk include by-products in the form of meat, rapeseed oil, straw and manure.

Table 3. *Crop rotations and average yield on the dairy farm studied in Paper II. Crop rotation 1 occupied 40 ha per crop, i.e. in total 7 x 40 ha, and crop rotation 2 14 ha per crop, i.e. 7 x 14 ha. The crop rotation was developed in cooperation with JTI, based on need for forage according to Olrog (2002) and crop yield in the County of Västra Götaland (Statistics Sweden, 2007)*

Yr	Crop rotation 1	Yield (kg/ ha &yr)	DM content	Crop rotation 2	Yield (kg/ ha &yr)	DM content
1	Spring barley	2440	86%	Spring barley	2350	86%
2	Ley 1	6000	29%	Grazing	6000	
3	Ley 2	6000	29%	Grazing	6000	
4	Ley 3	6000	29%	Grazing	6000	
5	Rapeseed	1693	91%	Grazing	6000	
6	Wheat	3228	87%	Grazing	6000	
7	Broad beans	2026	85%	Broad beans	2026	85%

The energy demand on the arable farm was dominated by tractor fuel for field operations and heat for grain drying, while the dairy farm had high electricity consumption due to automated milking and to manure management (pumping/stirring). The energy demand of both farms in the respective reference scenarios, based on diesel for field operations, oil for grain drying and electricity, is shown in Table 4.

Table 4. *Summary of energy consumption (MJ/yr) in the reference scenario on the arable farm (Paper I) and the dairy farm (Paper II). Heat and electricity for residential buildings was included only for the arable farm*

	Electricity	Heat	Hot water	Tractor fuel
Arable farm	50900	278400	12100	414000
Dairy farm	306500	115000	-	530300

4.2 Alternative energy systems on the arable farm – Paper I

Two energy self-sufficiency scenarios (ESS) were assessed for the arable farm in Paper I (Figure 2). The first scenario (ESS I) was based on biogas production from ley used as green manure in the reference scenario. The biogas was assumed to be used in an internal combustion engine for production of heat and electricity, and in an external upgrading plant for producing vehicle fuel. The upgrading facility used cryogenic technology to convert raw biogas to LBG quality, which was transported to pumping stations. The LBG was stored and transported in vacuum-insulated trucks, with short-term storage in vacuum-insulated tubes assumed to take place on the farm. The LBG had to be consumed relatively quickly in order to avoid leakage from these tubes.

Heat from the engine system was distributed between the farm buildings and the biogas plant. The electricity produced covered the requirement of the buildings, the biogas plant and the upgrading facility on an annual basis, although exchange with the national grid took place in order to buffer fluctuations in demand. Slurry from the biogas production was assumed to be returned to the fields to maintain the nutrient content.

The second scenario (ESS II) was based on energy extracted from wheat straw produced on the farm. The straw was assumed to be transported to a lignocellulosic ethanol production plant, where ethanol and electricity can be produced simultaneously. In the process, lignin bonds in the straw are destroyed and the cellulose and hemicellulose are converted to sugars via addition of enzymes and these sugars can be fermented to ethanol.

Electricity and process steam were assumed to be produced from the lignin separated in the process, and this electricity would in fact cover the farm and process requirements on an annual basis. There were assumed to be two straw boilers on the farm, one for heating of farm buildings and one, with higher capacity, for grain drying.

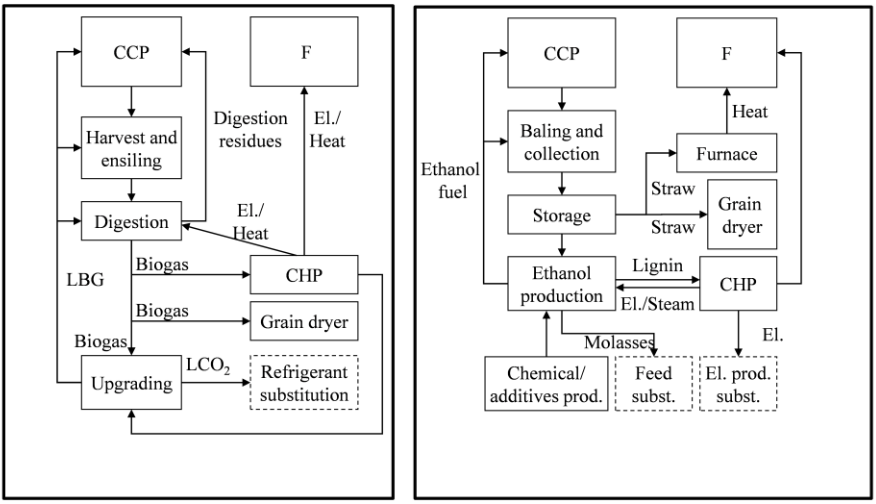

Figure 2. Energy and material flows in the self-sufficiency scenarios on the arable farm in energy self-sufficiency scenario (ESS) I (left) and ESS II (right). Cash crop production is referred to as CCP and the farm as F. CHP is combined heat and power production. Dotted boxes represent processes avoided in the self-sufficiency scenarios.

4.3 Alternative energy systems on the dairy farm – Paper II

Two energy self-sufficiency scenarios were considered in Paper II, referred to as the biogas system and an RME system.

In the biogas system, biogas was assumed to be produced from a mixture of manure and straw in an anaerobic digester, dimensioned to process all manure produced on the farm and with addition of straw in order to cover the entire energy demand. The raw biogas was divided into two streams. The first stream was conducted unprocessed to a gas engine stationed on the farm, dimensioned to produce electricity meeting the farm's needs. The second stream was led via a pipeline system to an upgrading unit, using the same cryogenic technology as in Paper I. Heat for grain drying was supplied by a straw boiler.

In the RME system, both biogas and RME were produced. The substrate available at the farm (manure with mixed-in bedding material, *i.e.* straw) was used for biogas production in an anaerobic digestion chamber and for electricity production in a gas engine. Rapeseed oil, a by-product from rapeseed cake (protein feed) production, was assumed to be utilised in a small-scale system on the farm to produce the RME used as tractor fuel. Hence, the production system was completely farm-based and storage also took place on the farm. Heat for grain drying was assumed to be supplied by a batch-fired straw boiler.

4.4 Results

The results showed that both the arable farm and the dairy farm studied can become self-sufficient in energy. On the arable farm (Paper I), ESS I required ley to be harvested from 25 ha, which was 13% of the total farm area. ESS II required 49 ha of straw, which was 25% of the total farm area. This biomass was available, since ley was grown on 29% of the farm area and cereals on 44% in the given crop rotation.

On the dairy farm (Paper II), the RME scenario required all available rapeseed to be used. In the biogas scenario, straw from an additional 49 ha compared with the RME scenario was required to cover the entire farm demand.

4.4.1 Energy balance

In the bioenergy-based scenarios, the energy demand was typically higher due to process energy. The ethanol-based scenario, ESS II, on the arable farm required more energy than ESS I, but also utilised a larger fraction of the energy content of the biomass in the form of electricity production from lignin (Paper I). A considerably larger fraction of the energy used was process energy for the conversion to energy carriers in this scenario compared with ESS I (Table 5).

Table 5. *Energy requirement as total energy consumption (TEC, GJ), energy carrier production (ECP) as a fraction of TEC, ECP as a fraction of energy consumption for cash crop production (CCP) and farm buildings (F) and the fraction of energy in the biomass that was utilised (Paper I).*

	TEC	ECP/TEC	ECP/(CCP+F)	Fraction of energy in biomass converted to energy carrier			
				El.	Heat	Fuel	Tot
ESS I	1020	24%	31%	4%	16%	20%	40%
ESS II	1416	45%	83%	7%	23%	27%	57%
Reference	755	24%	24%	n/a	n/a	n/a	n/a

There was a considerable increase in heat demand on the dairy farm (Paper II), due to the biogas production and somewhat higher electricity use in the alternative scenarios, for biogas and RME production, compared with the reference scenario (Table 6). Biogas production required process heat in the order of 15-20% of the energy contained in the gas.

Table 6. *Change in energy use due to energy self-sufficiency on the dairy farm. Additional energy use is based on renewable resources on the farm (Paper II)*

Reference scenario	GJ/yr	Biogas scenario	GJ/yr	RME scenario	GJ/yr
Heat	115	Heat	350	Heat	342
Electricity	307	Electricity	403	Electricity	362
Fuel	701	Fuel	699	Fuel	699
Total energy	1123	Total energy	1452	Total energy	1403

4.4.2 GHG emissions

The reference scenario for the arable farm is shown as the baseline in Figure 3 and Table 7, whereas it is presented separately for the dairy farm. On the arable farm, the GHG emissions reduction was 35% in the biogas-based scenario (ESS I) and 9% in the ethanol-based scenario (ESS II) (Paper I). Figure 3 shows the GHG emissions relative to the reference scenario, which constituted the baseline.

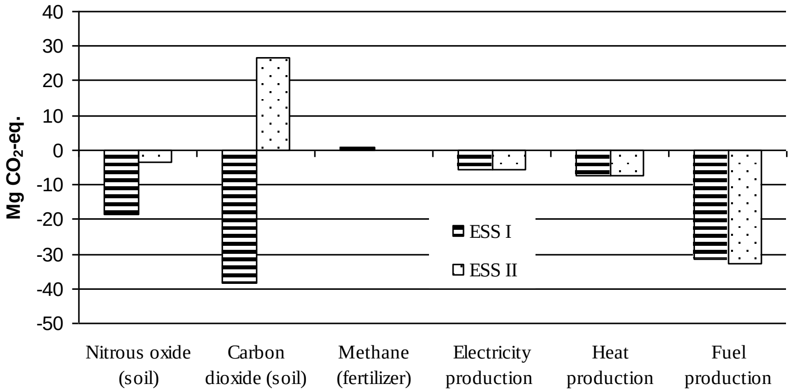

Figure 3. Disaggregated GHG emissions per FU from the systems on the arable farm relative to the baseline, *i.e.* the reference scenario based on fossil fuels (Paper I).

Nitrous oxide emissions from the soil were lower in both self-sufficiency scenarios compared with the reference scenario, but the impacts on soil carbon content differed significantly. Methane released from digestion residues produced in ESS I had only a minor impact on the results, while production of all energy carriers, in particular fuel production that conveyed useful co-products, gave considerably lower GHG emissions than energy use in the reference scenario.

On the dairy farm, the GHG emissions reductions were 43% in the biogas system and 32% in the RME system (Table 7).

Table 7. *GHG emissions per FU, by source, on the dairy farm (Paper II)*

	Reference	Biogas	RME
Fossil fuel use	155		
RME production			1
Production of biogas plant		3	3
Production of CHP plant		14	14
Production of straw boiler		1	1
Digging trenches for pipelines		7	
Field operations	88	0	0
Biogas upgrading losses		3	
RME production			1
Soil-C, Crop rotation 1	90	13	83
Soil-C, Crop rotation 2	-19	-21	-21
N_2O from fertilisation and crop residues	180	176	173
N_2O from carbon losses	31	20	32
Manure management			
- methane	114	6	3
- N_2O	6	6	6
Enteric fermentation	493	493	493
Substitution			
Electricity substitution			-5
Diesel substitution			
Substituted meat production	-125	-125	-125
Substituted oilseed production	-50	-50	
Total (g CO_2-eq./kg FU)	962	545	687

Enteric fermentation was the largest emissions item and was unchanged in the scenarios. Passing manure through AD resulted in a significant reduction in methane emissions to the atmosphere from manure management, as methane was collected and combusted.

Digestion of manure and straw also had a positive effect on soil carbon content, as the processed substrates had a faster humification rate, *i.e.* were sequestered in the soil to a larger extent instead of being mineralised to carbon dioxide.

5 Part 2: Local energy supply

5.1 Basic system descriptions

In Part 2, the area of the farms supplying the energy systems was not defined, but was calculated as a function of the energy demand of the villages and energy efficiency of the conversion processes. The supplier could be a farm cluster rather than one single farm.

The farm(s) in Paper III utilised the same crop rotation as the arable farm in Paper I, including an N-fixing ley crop twice in the seven-year rotation (Table 1). The green manure was assumed to be ploughed back into the soil in the scenarios in which it was not used as substrate for biogas production. It was also assumed that the farm had set-aside land available, which in the reference system was unmanaged (not fertilised or harvested) grassland that could be used for Salix production.

The systems in Paper III produced heat and supplied electricity for a newly built village of 150 modern households. The electricity grid was used as a buffer, but on an annual basis the same amount was produced and consumed. In the reference system, heat was supplied by heat pumps and all electricity required was produced in a large-scale natural gas power plant.

In Paper IV, the farms had both agriculture and forestry, but were no longer assumed to be organic. The systems produced district heating for a village with 425 households (1000 inhabitants), with the same energy standards as in Paper III. The reference scenario was based on heat produced in large-scale biomass plants located 20 km away in a larger city.

In Paper III, the FU was CO_2-equivalents associated with the entire heat and power supply to the village for one year. In Paper IV, the FU was CO_2-eq./MJ of heat supplied to the end-user.

5.2 Alternative heat and power systems – Paper III

Three energy supply scenarios were assessed in Paper III and are referred to in that paper as Bio 1-3. However, the exact same denomination was used in Paper IV, so in the following text the scenarios in Paper III are referred to as A, B and C (Figure 4).

In scenario A, ley (green manure) was harvested and used as substrate for biogas production. The biogas fuelled a microturbine for production of heat and electricity. Straw was used for additional heat production.

In scenario B, Salix produced on the set-aside land was chipped and fed through a gasification chamber. The gas produced was used for combustion in an internal combustion engine for heat and electricity, with 40% electric efficiency and 90% total efficiency.

In scenario C, Salix was chipped and combusted in a boiler, fuelling a Stirling engine via a heat exchange system. The engine was assumed to have 25% electric efficiency and 90% total efficiency.

Straw boilers (using wheat straw) were used in scenario A and B to make up for insufficient heat output in the winter time from the CHP systems. The thermal efficiency of the straw boiler was assumed to be 90%.

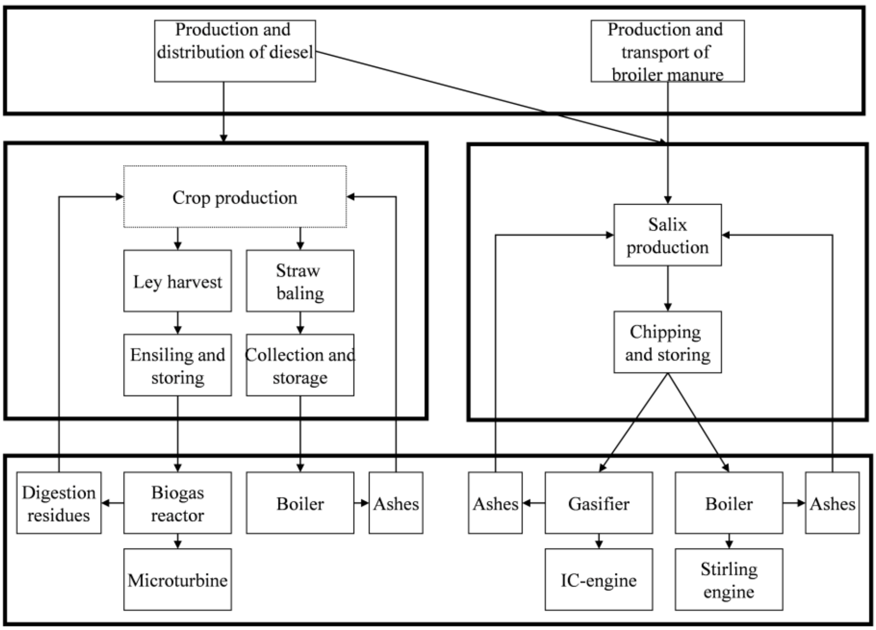

Figure 4. Schematic diagram of the biomass supply chain in Paper III. Ley was used in scenario A, straw in scenario B and willow chips in scenarios B and C. The full lines separate physical production areas. Crop production was the reference position from which changes induced by biomass production for energy generation were calculated.

5.3 Alternative heat systems, ownership and heat price – Paper IV

There were four bioenergy scenarios in Paper IV, in which the farm cluster was integrated vertically into the production chain to different levels (Table 8).

Table 8. *Summary of the reference scenario and Bio 1-4 in Paper IV, showing the production system (left) and whether the energy utility or farm cluster was responsible (right)*

	Production system	Fuel supply	Production	Distribution	Sales
Reference scenario 1	Natural gas boiler working on flexible loads, both summer and winter	Utility	Utility	Utility	Utility
Bio1	Culvert drawn from a town 10 km away. Heat produced by biomass CHP. Regionally sourced forest residues[a]	Utility	Utility	Utility	Utility
Bio2	Regionally sourced forest residues[a]. Pellet for peak/low loads[b]	Utility	Utility	Utility	Utility
Bio3	Local forest residues. Pellet for peak/low loads[b]	Cluster	Cluster	Utility	Utility
Bio4	50% local forest residues[c] and 50% Salix. Straw for peak/low loads. Storage on the field [c]	Cluster	Cluster	Cluster	Cluster

[a]Average transport distance 50 km
[b]Average transport distance 150 km
[c]Average transport distance 10 km

Forest residues were assumed to be harvested at a rate of 14.7 tonnes DM/ha and year (Petersson, 2013), from thinnings and full harvests. Straw yield, a by-product of grain production, was assumed to be 3.2 tonnes DM/ha and year on agricultural land.

The average yield of a Salix rotation (22 years including recovery) was 3.9 tonnes DM/ha and year, with harvesting taking place every 3-5 years. Diesel consumption for each feedstock type was calculated (Table 9).

Table 9. *Diesel consumption for biofuel production (Source: Petersson, 2013; Gonzalez-Garcia* et al.*, 2012; Ahlgren* et al.*, 2009)*

	Forest residues – local (L/dry tonne)	Forest residues – regional (L/dry tonne)	Salix (L/dry tonne)	Pellets (L/tonne)	Straw (L/tonne)
Raw material production	n/a	n/a	6.9	n/a	0
Forwarding to roadside	2.2	2.2	n/a	n/a	3.1[a]
Chipping	2.8	2.8	2.8	n/a	0
Transport to heat plant	0.2	1.1	0.2	3.0	1.3
Handling at the plant[b]	0.5	0.5	0.5	n/a	n/a
Transport and spreading of ash	0.1	0.1	0.0	0.0	0.1
Total	5.7	6.6	10.4	3.0	4.5

[a]Includes rowing and baling
[b]Mainly use of frontloader at the plant

Production costs (Table 10) were broken down into six cost components, in accordance with the system used for reporting of financial data to the Swedish Market Energy Inspectorate (EI). The average profit margin for medium-scale and small-scale utilities was used for the energy utility and farm cluster, respectively, to estimate a cost-driven heat price in each fuel supply system. The cost component raw material (fuel) was replaced with a detailed calculation of fuel costs in each scenario (the reference scenario and Bio 1-4).

Table 10. *Average heat production (GJ/yr), specific production costs (SEK/GJ) and profit margin (%) for segments of different sizes in the district heat industry in Paper IV*

	Average heat sales in each category	C_1: Raw materials[a]	C_2: Personnel[b]	C_3: Other external costs[c]	C_4: Other variable costs[d]	C_5: Capital costs[e]	C_6: Profit margin[f]
	GJ/yr			SEK/GJ			
Small suppliers	113 437	104	12	42	2	36	12%
Medium-sized suppliers	535 807	91	15	34	1	27	13%

[a]Fuels and other external purchases (excl. input energy), here replaced with detailed fuel cost calculations
[b]Costs for own staff
[c]Manning and other external costs for operation and maintenance
[d]Other unidentified or uncategorised costs
[e]Write-offs and write-downs
[f]Profit margin; earnings before interest and taxes (EBIT) divided by total heat sales

5.4 Results

In Paper III, scenario A appeared to require rather a large area (Table 11), but since the ley crop was integrated into the crop rotation for fertiliser purposes, it does not compete with food production. However, the area required meant that the farm was rather large (1061 ha) in Swedish terms, since the ley crop only occurs in the third and sixth year of the rotation.

The straw requirement was comparatively large due to the low heat output of the CHP plant, and therefore a straw boiler of higher capacity than that in scenario B was required.

The requirement for set-aside land was significantly higher in scenario C than in scenario B, due to the low electric efficiency of the CHP plant in the former scenario. This resulted in large amounts of energy not being utilised and therefore lost as excess heat in order to reach the required electric output.

However, only 40 ha of straw were required in scenario B and 0 ha in scenario C, which meant that the food production area required was significantly reduced compared with scenario A.

Table 11. *Land requirement (ha) in scenarios A-C (Paper III)*

	Scenario A	Scenario B	Scenario C
Farmland			
Ley	303	0	0
Straw	99	40	0
Set-aside land			
Salix	0	81	169
Total	402	121	169

In Paper IV, the yearly land requirement (Table 12) was calculated for each fuel and bioenergy scenario except wood pellets, which are made from sawmill residues and have no land requirements of their own. The forested land area was not considered occupation of land, however, as the biomass used as fuel is a by-product from timber and pulp & paper production. These data are shown simply an indication of the forest and agricultural resources required (Table 11).

Salix is somewhat different because it occupies agricultural land that could have been used for other purposes. The value given in Table 12 refers to the total area of Salix plantation required, not the area harvested each year.

Table 12. *Land requirement (ha/yr) in the bioenergy scenarios in Paper IV. The Salix plantation requires additional land compared with the reference scenario, whereas forest residues (FR) and straw were harvested from existing forest and agricultural fields*

	Bio 1	Bio 2		Bio 3		Bio 4		
	FR (regional)	FR (regional)	Pellets	FR (local)	Pellets	FR (local)	Salix	Straw
Forest land	101	80		80		20		
Cropland							105	76

5.4.1 Energy balance

The energy use in Paper III was calculated in two ways; fossil energy requirement (FER) per FU and primary energy requirement (PER) per FU. Annual biomass consumption of each system and the amount of excess heat produced were also calculated (Table 13).

Table 13. *Fossil energy requirement (FER), primary energy requirement (PER) and biomass requirement in each scenario in Paper III. Excess heat produced in each of the scenarios is also shown*

Scenario	FER (GJ/ FU)	PER (GJ/ FU)	Biomass (Mg/ FU)	Excess heat
A	600	35 920	2 100	< 1 %
B	270	14 000	760	13 %
C	490	25 000	1 360	140 %
Nat. gas	11 640	12 800	0	0

In Paper IV, the primary energy (PE) factor for heat produced by each respective fuel type was calculated as the sum of the energy contained in the fuel (lower heating value, LHV) and the energy required in the production phase. The input energy demand was also broken down into energy carriers and calculated as percentage of the energy (LHV) contained in the fuel (Table 14).

Heat for pellet production was supplied by biomass residues (bark), whereas according to CLCA practices electricity was assumed to be marginal and based on natural gas. The PE factor for natural gas is 1.09 (Gode *et al*., 2011), including all energy required for extraction and transmission.

Table 14. *Primary energy (PE) factor and input energy per fuel and energy carrier (Paper IV)*

	Natural gas	Forest residues (regional)	Forest residues (local)	Pellets	Salix	Straw
PE factor	1.09	1.016	1.014	1.235	1.026	1.011
Diesel[a]		1.6%	1.4%	0.7%	2.6%	1.1%
Electricity				1.8%		
Bioenergy				20%		

[a]Includes production stages, transport to heat plant and ash handling

5.4.2 GHG emissions

In Paper III, GHG emissions in scenario B and scenario C were 60 and 42 Mg CO_2-eq./FU, respectively. However, the most notable finding was for scenario A, where the net GWP was negative (-19 Mg CO_2-eq./FU). This was due to a cultivation system where ley was digested instead of being ploughed in fresh, which increased the soil carbon content and reduced emissions of nitrous oxide from the soil. In the natural gas scenario, net emissions were 351 Mg CO_2-eq./FU (Figure 5). Soil emissions dominated the GWP category for scenarios A-C. The emissions were broken down into emissions of carbon dioxide and nitrous oxide, both expressed in CO_2-equivalents.

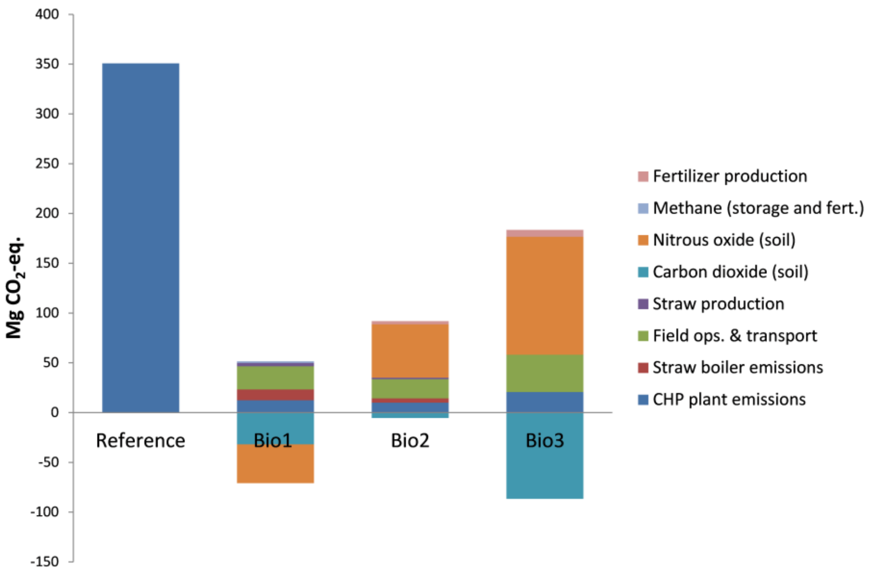

Figure 5. GHG emissions (Mg CO_2-eq.) for one year supply of heat in each scenario in Paper III. From left to right: Natural gas and scenarios A, B and C.

The carbon uptake of the soil also increased in scenarios B and C, where a carbon sink was created due to the establishment of SRC on set-aside land. The effect was larger in scenario C because the area planted was about twice as large as in scenario B (Paper III).

On the other hand, emissions of nitrous oxide increased compared with the reference case, due to fertilisation of the set-aside land (unfertilised in the reference scenario) and the higher quantities of crop residues on the fields (Paper III).

In Paper IV, the reference scenario with natural gas-based heat production emitted 107.3 kg CO_2-eq./GJ delivered heat, while that for the biomass scenarios ranged between 3.3 and 14.3 kg CO_2-eq./GJ (Table 15). Negative emissions were obtained when Salix was mixed with forestry residues for base production in Bio 4, because the carbon sequestration effect in the soil exceeded the impact from the fossil input to Salix production.

Table 15. *GHG emissions per unit of fuel (kg CO_2-eq./GJ fuel) and per scenario (kg CO_2-eq./GJ delivered heat) in the reference scenario (Ref) and in scenarios Bio 1-4 (Paper IV)*

	Ref	Bio 1	Bio 2		Bio 3		Bio 4		
	Natural gas	FR regional	FR, regional	Pellets	FR, local	Pellets	FR, local	Salix	Straw
Diesel use		1.1	1.1	0.5	1.0	0.5	1.0	1.8	0.7
Electricity use[a]				5.6		5.6			
SOC		6.7	6.7		6.7		6.7	-11.6	2.5
Combustion	82.8								
Added N									0.9
Total kg CO_2-eq./GJ fuel	82.8	7.8	7.8	6.1	7.7	6.1	7.7	-9.8	4.2
Total kg CO_2-eq./GJ heat delivered	107.3	14.3	13.3		13.1		3.3		

[a]Includes production and transmission of electricity

5.4.3 Cost analysis

The cost-based heat price calculated in Paper IV indicated that the more integrated the farm cluster was in the value chain, the lower the estimated heat price (Table 16). Straw as peak fuel was able to reduce the cost of fuel considerably, in particular compared with natural gas, but also regionally sourced forest fuels.

With a fully integrated farm cluster the price was halved compared with a fossil reference, a difference which was mainly due to the energy and CO_2 tax in Sweden, from which biomass is exempt.

Table 16. *Cost-driven heat price (SEK/GJ produced heat) in the reference scenario (Ref) and in scenarios Bio 1-4 (Paper IV)*

	Ref	Bio 1	Bio 2	Bio 3	Bio 4
C1: Raw material	234	93	94	95	80
C2: Personnel	18	18	18	12	12
C3: Other external costs	36	36	36	42	42
C4: Other variable costs	5	5	5	2	2
C5: Capital costs	26	26	26	36	36
C5: Profit margin	14%	14%	12%	12%	12%
Raw heat price (SEK/GJ)	363	202	200	209	193
Fuel cost as % of total cost	64%	46%	47%	45%	42%

6 Discussion

6.1 Energy self-sufficiency at organic farms

The energy self-sufficiency potential of agricultural farms was investigated in Part 1 of this thesis. The results showed that from a technical and resource perspective, both arable and dairy organic farms can become self-sufficient in energy by using their own on-farm agricultural residues.

Fredriksson *et al.* (2006), Hansson *et al.* (2007), Halberg *et al.* (2008) and Ahlgren *et al.* (2009) have all previously shown that self-sufficiency in tractor fuel or for heat and power on organic farms is possible. They also show that this can be achieved with a reduced climate impact compared with using fossil fuels. GHG emissions reduction potential in those studies was roughly between 50-95%, depending on the raw material and technology used.

Somewhat different GHG emissions reduction potential values were obtained in the present thesis. For the arable farm in Paper I, the GHG reduction obtained in the straw-based scenario was only 9% compared with a fossil system, but was 35% in the biogas-based scenario. The discrepancy between results can be explained by the fact the previous studies cited above, like most early LCA studies of bioenergy, did not account for soil carbon changes (see section 6.3).

Guan *et al.* (2014), modelled energy use on a dairy farm and its potential to become energy self-sufficient by heat and electricity production from biogas and found that the farm could become self-sufficient as the energy contained in the manure produced exceeded (by some margin) the net energy demand.

Paper II showed that the GHG emissions from production of 1 litre of milk can be reduced by up to 43% by replacing fossil fuels with biogas produced from manure and straw. For comparison, a recent study by Battini *et al.* (2014) on milk production in Italy comparing a fossil-based energy system to a system where manure was used to produce biogas for heat and power on the farm found that GHG emissions from 1 litre of milk were reduced by 24% in the biogas scenarios compared with the reference, or 37% assuming that slurry was stored in an airtight tank (as was also assumed in Paper II).

There is in fact a rather large number of LCA studies on the climate impact of milk production. Most of them report a figure of between 0.3 and 2 kg CO_2-eq./kg ECM and particularly in the range 1-1.5 kg CO_2-eq./kg ECM (*e.g*. Roer *et al*., 2013; Flysjö *et al.*, 2011; Thomassen *et al*., 2008; Cederberg *et al*., 2007). This is similar to the results of the base scenario in Paper II in this thesis, with results of just under 1 kg CO_2-eq./kg ECM in the reference (fossil) scenario.

It is clear from these studies that methane from enteric fermentation and manure management remains the largest contributor of GHG emissions from dairy production, around 50% of total GHG (see Table 7). To lower emissions from enteric fermentation, for example by a change in diet or breeding, may hence be as important, or more important, as replacing fossil fuels in the agricultural sector.

6.2 Local heat and CHP systems

Part 2 of this thesis showed that decentralised heat systems for rural villages are favourable to heat or CHP produced based on natural gas. LCA studies with similar conclusions have been produced before, *eg* Muench and Guenther (2013) and Eriksson *et al.* (2007).

However, Paper III and Paper IV also show that the heat or CHP systems are feasible based on locally produced agricultural fuels, *i.e.* crop residues and SRC, with lower GHG emissions and (shown in Paper IV) lower price than regionally sourced forest residues or pellets. While shorter transport distance plays a relatively small role for these results, the impact on soil carbon dynamics is a critical parameter (see Figure 5 and Table 15). Short rotation coppice as well as digestion of ley (used as green manure) will have a soil carbon build-up effect whereas straw removal will have a carbon depletion effect (see section 6.3). Since removal of harvest residues from the forest floors have a corresponding impact to that of crop residue removal, *i.e.* decrease soil carbon content, there is in fact a double carbon sequestration effect when replacing forestry fuels with Salix chips.

At the same time, from an economic perspective straw constitutes a highly competitive alternative to forestry residues as long as the transport distance is low. According to Glithero *et al.* (2012), if the purpose is to maximize the gross profit margin at the farm, straw should be baled and sold to the market (this conclusions is however clearly dependent on the availability of a market and the market price).

On the other hand, SRC, *i.e.* Salix in these scenarios, is more expensive than forestry residues to produce, and in fact less suitable for Swedish heat and CHP plants, that are typically adapted to forestry fuels. According to Rosenqvist *et al.* (2012) the price of Salix wood chips can be reduced by up to 35% if the cultivation were to be expanded to generate economies of scale, and taking into account learning effects. Further, the production costs depend to a rather large part on uncertain and fluctuating establishment costs as well as opportunity costs, which in turn depend on the global market price of wheat (Rosenqvist *et al*., 2012).

The competitiveness could also potentially increase if the carbon sequestration effect of Salix plantations were recognised as an environmental service and internalised into the market price, or encouraged via financial incentives through the CAP mechanisms.

6.3 Soil carbon dynamics and crop residue removal

At the time when this thesis was initiated in 2008, the impact of different cropping systems on soil carbon dynamics was not often included in LCA studies. It was generally considered to be very difficult to account for as there are many soil parameters involved that can be rather site-specific, for example soil type, water content, C/N ratio and exposure to oxygen (Kätterer *et al.,* 2011).

Over the last few years, the models and methodologies for analysing soil carbon dynamics have been refined and increasingly included in bioenergy LCA. An adapted version of the ICBM model was introduced early for the work in this thesis, and was consistently used to model soil carbon dynamics for the crop rotation on the farm, with a varying degree of straw removal and spreading rate of manure/digestion residues.

A major increase in bioenergy use is projected world-wide and also in the EU (Gabrielle *et al*., 2014; IEA, 2013; IPCC, 2013; Cherubini & Ugliati, 2010), and a large proportion of this is assumed to originate from crop residues. Residue use is incentivised by the EU RED, which gives residues zero emissions in assessments of GHG. This thesis showed that the impact of biomass-based energy systems on SOC can have a major impact on the net GHG emissions reduction potential of biomass systems, as illustrated by *e.g.*

Figure 3 and Table 7. Similar results have been reported by *e.g.* Holma *et al.* (2013), who found that Fischer-Tropsch diesel only fulfils the upcoming sustainability criteria of a 60% GHG emissions reduction (due in 2018) under the RED if the changes to soil carbon stocks caused by feedstock production are excluded.

Crop residue retention in fields can be considered a global warming mitigation method in itself, often referred to as soil carbon sequestration, and has the additional benefits of retained nutrient content, improved soil structure, better water infiltration into the soil and reduced risk of erosion (Lal, 2008). A long-term trial in Uppsala, Sweden, found that topsoil carbon concentration had decreased most, by more than 33%, in a soil lying fallow for 53 years compared with other treatments, and had decreased least when covered with a perennial low-tillage crop such as a SRC or ley (Kätterer *et al.*, 2011). Whittaker *et al.* (2014) found that soil carbon losses stemming from straw removal range between 0.39-2.35 t CO_2-eq./ha and year. According to Lal (2008), an increase in soil carbon of 1 tonne/ha and year by crop residue retention can potentially increase global grain yield by 20-40 million tonnes per year. The implications of crop residue removal are clearly important.

Salix can contribute to SOC via fine roots, leaves and litter, and there is in fact quite strong scientific support for a soil sequestration effect of Salix plantations (Ericsson *et al.*, 2014; Rytter *et al.*, 2012). Hammar *et al.* (2014) used the ICBM model to simulate various cropping systems with different yields and over different time periods, and found that a Salix plantation standing for 100 years would give -16 Mg CO_2-eq./ha assuming a yield of 20 t DM/ha from the first harvest and 30 from subsequent harvests (the field is harvested every 3 years). However, the results are dependent on yield; low yields of 10 and 17 tonnes DM/ha for the first and subsequent harvests, respectively, resulted in net release of carbon to the atmosphere. In Paper II of this thesis, this soil sequestration potential of Salix plantations gave the somewhat counterintuitive effect that a system with low electric efficiency had a lower climate impact than technically more efficient systems because it required more extensive SRC plantations.

There is also evidence that crop residue removal in fact can *increase* nutrient availability to the following crop, as *e.g.* cereal straw can immobilise mineralised nitrogen when decomposing. However, straw contributes to a higher nitrogen content and fertility in the long-term perspective (Powlson *et al.*, 2011).

In the arable farms studied in Papers I and III, the organic farms used green manure systems that rely on ploughed-in ley to supply nitrogen. Such systems can benefit from harvesting and processing the ley via anaerobic digestion rather than ploughing in ley fresh. The digestion residues can be more precisely matched to the requirements of the crop growth cycle, resulting in lower emissions of N_2O from the soil (Möller & Stinner, 2009; Stinner *et al*., 2008). Indirectly, this could potentially also contribute to higher carbon content in the soil through higher yields, due to the better nitrogen availability.

Furthermore, on dairy farms such as that described in Paper II, spreading digestion residues on the fields instead of fresh manure as a nitrogen fertiliser can have a positive effect on soil carbon content as the digested material is more stable, meaning that a lower fraction of organic carbon is mineralised to CO_2 (Marcato *et al*., 2009). This effect is also reflected in the humification coefficients included in the ICBM model used in this thesis. The exact values for these coefficients in soil carbon dynamic models have however been subject to discussions and modifications over time, and there are different methodologies for scientifically establishing the coefficients. These include long-term field experiments, incubation studies (*i.e*. laboratory-scale experiments) or estimates based on biochemical properties, such as the Biological Stability Index (Kätterer *et al*., 2011). The assumption of two or more carbon pools is also a simplification and the decomposition rate in the pools depends on local conditions such as soil moisture, temperature conditions, topsoil cover, ratio of above-ground and below-ground biomass input, C:N ratio *etc*. (Kätterer *et al*., 2011; Grogan Matthews, 2002).

6.4 Biogenic carbon

A concern for LCA practitioners in the bioenergy field is whether biomass combustion systems really can claim to be climate neutral. Carbon neutrality means that the same amount of C as is released during combustion of biomass is bound by the next generation of the crop, and can be true as long as forests are managed in a sustainable manner and following crops grow with equal (or higher) yield.

However, this does not necessarily mean climate neutrality, as the carbon molecules released during biomass combustion still linger for a while in the atmosphere before they are bound by a new generation of crops, *i.e*. there is a time dependency of the climate impact (Pawelzik *et al*., 2013; Ericsson *et al*., 2012). In LCA studies, it is often simply assumed that biomass-based energy systems create a sustainable carbon cycle that makes the system both carbon and climate neutral.

Zetterberg & Chen (2011) compared the energy balance in a radiative forcing model when tops and branches from forestry were harvested and instantly combusted, compared with when they were left to mulch (hence slowly releasing carbon to the atmosphere). They found that bioenergy systems based on forestry residues are not climate-neutral due to the time dependency. The model also accounted for the contribution to SOM that would result from leaving the residues to mulch. Ericsson *et al*. (2013) developed a model simulating how released GHG emissions slightly alter the energy balance (*i.e*. temperature) of the Earth due to radiative forcing, by including an impulse response function (IRF). With the development and refinement of such models, the biogenic carbon parameter can now be included in bioenergy LCA to obtain more accurate results.

For this thesis, including the effects of time dependency would probably have shifted the results slightly less in favour of some of the scenarios, such as the straw-based systems, but more in favour of the scenarios based on SRC, due to the temporary sequestration of carbon in standing biomass during the 3-4 years between harvests, and in roots during the whole 20-year rotation.

6.5 Consequential LCA as an assessment method

CLCA was chosen over ACLA for the studies in this thesis, as it was deemed to more accurately reflect the impact of a change – here the introduction of new heat, power and/or fuel production capacity.

There is however a potential weakness of CLCA, in the inherent uncertainty when the market response (*i.e.*, which production capacity that is affected by an increase or decrease in demand of a product or service) must be predicted, either by dynamic economic models or assumptions based on detailed market knowledge. System boundaries and assumed substitution depend on how markets react, which means taking into account market behavioural factors. Furthermore, production and demand are not always elastic, and that there may be constrained suppliers or markets, which means that markets are in fact not affected by the change in demand. There is also a scale and time dependency of the market response and if the product system analysed is very small compared with the market as a whole, it might only affect marginal upstream production processes. (Rebitzer *et al*., 2014)

Despite continuous work towards making LCA a more robust and transparent methodology, there is still little standardisation regarding how to handle such uncertainties (*e.g*. Plevin *et al*., 2014), especially in CLCA (Curran, 2013; Pawelzik *et al*., 2013). Some kinds of uncertainties apply specifically to CLCA, but there are also uncertainties due to choice of reference system, time frame, site specificity, time specificity, or simply lack of validated data or field experiments in both ALCA and CLCA (Holma *et al*., 2013). Uncertainties become an issue when LCA results are being applied by decision- and policymakers, creating a need for close monitoring of the impacts arising from environmental improvements through corporate or political decisions.

The difference between ALCA and CLCA, and why they apply for different purposes has been debated by LCA practicioners. Plevin *et al.* (2014) argue that ALCA can be misleading to policy makers as it does not estimate the effect of a change and leaves out critical elements such as market response. The authors recommend use of CLCA and point out that its accuracy can be improved by comparing various plausible scenarios to each other, hence “cancelling out” uncertainties that apply to all scenarios. Dale & Kim (2014) responded that there is no proof that CLCA would provide a more accurate prediction of the real world than ACLA. Furthermore, the authors argue that most LCA studies are in fact hybrids, with a very small element of market analysis or CLCA to them. Brandão *et al*. (2014) argues that ALCA may be more *precise*, but CLCA more *accurate* – meaning that even if there may be more uncertainties in CLCA than ALCA, the latter may be more inaccurate if uncertain, but critical, parameters are omitted in the ALCA.

Anex and Lifset (2014) made a comparison to cost accounting methods in business management, which over time have shifted from cost management to cost inventory accounting, *i.e*. more towards an attributional rather than consequential approach, basically in order to facilitate auditing. However, they recognised that the inventory-based cost accounting method does little to help firms understand the opportunity costs of decisions and actions (Anex & Lifset, 2014).

Plevin *et al*. (2014) raised the issue of indirect or scale effects of introducing a new production system on a market, *e.g*. if a biofuel were to gain a considerable market share in a certain region and replace use of petrol or diesel. This could lead to a drop in the price of petroleum products and hence encourage use of petroleum elsewhere (since the oil market is global), or in other sectors such as power production. The important point here is that there is no perfect substitution between one product and another; in fact, implementing a low-carbon regime by forceful renewable energy policies (such as emissions caps, CO_2 taxes or quota system for renewable energy use) in one part of the world may increase use or production of emission-intense products/processes in another region. This mechanism is often referred to as carbon leakage. As Hertwich (2014) concludes, it is not biofuels and other renewable energy systems that mitigate climate change *per se,* but the fact that fossil fuels can be left in the ground as a consequence of the dispersion of new technologies that does so.

Many LCA studies (both CLCA and ALCA) today apply the simplified GWP_{100} metric. The physical principles behind the warming effect on the planet are however complex and involve parameters such as radiative forcing, temperature gradients and discount rates. There is clearly no absolute truth to GWP_{100} (Plevin *et al*., 2013); in fact the GWP_{100} metric has been discussed both with respect to GWP values (the potency of different greenhouse gases relative to CO_2) and the time perspective, 100 years retention time in the atmosphere. It has been recognized by IPCC that the emission metrics are associated with uncertainties and value judgement, such as how the climate effect is judged, weighting of climate effects, incorporation of physical and economic considerations *etc*. The IPCC now recommends that the policy context is taken into account to determine the time horizon (IPCC, 2014).

The comprehensive approach and the inclusion of an entire life cycle is the strength of CLCA; it helps pinpoint emissions hotspots and reduce the risk of these being transferred to other parts of the supply chain by partial, sub-optimising assessments. Even if CLCA does not provide perfectly accurate answers to all the questions associated with mitigation of climate change and the climate impact of bioenergy, it has contributed to identifying and quantifying key issues related to climate change and its mitigation, such as iLUC and the importance of soil carbon dynamics (for biomass systems in particular), carbon leakage and how to take into account the actual multi-market responses to corporate and policy decisions.

6.6 Marginal electricity production in CLCA

In this thesis, natural gas in condensing power plants was assumed throughout to be the long-term marginal power production technology. This assumption was based on an assessment of planned investments in electricity capacity made in 2007 (Kjaerstad & Jonsson, 2007).

However, in 2013 investment in wind and solar power production was in fact growing at a faster rate than investment in any other power technology, including gas-fuelled condensing plants (REN21, 2014). This development could change the prerequisites for determining long-term marginal electricity production, which perhaps should be a renewable energy source.

Moreover, the EU has set a cap on GHG emissions in Europe via its emission rights trading system (EU ETS), which in an ideal world would mean that new power capacity should not add to total net GHG emissions. Assuming that this policy works perfectly in practice, long-term marginal power production should be nearly CO_2-free (Finnveden, 2008). The EU ETS system has not worked as intended, however - there has been an over-supply of emission certificates on the market, resulting in plummeting prices and not the intended steering effect (EC, 2012).

It is a rather common belief that future renewable power systems will built on a variety of fuels and technologies, including wind, solar hydro, biomass and perhaps emerging technologies such as wave power, hydrogen-based technologies *etc*. There is in any case no single technology or resource that will meet the entire global power demand. Mathiesen *et al*. (2009) argue that a as a realistic market response to an increase/decrease in energy demand is a change in production capacity for a set of technologies rather than one single technology, and there are various dynamic models based on technical, economic and policy data that have been developed to predict this marginal energy mix.

There is however obviously an inherent major uncertainty in such models, as they attempt to project future scenarios. Several authors have discussed this uncertainty and propose a solution, *e.g*. Soimakallio *et al*. (2011) recommend that sufficient scenarios be included in CLCA to cover the range of typical emissions from electricity production. In this thesis, sensitivity analyses were conducted in all papers to analyse the impact on the results if power production had been based on wind power or coal condensing production instead of natural gas.

The future energy system is also projected to include more decentralised production and less central, large-scale production units (IEA, 2014b). This is facilitated by the development of smart grids, which can handle a larger amount of power sources than the conventional transmission and distribution networks. If this represents the future, including a mix of technologies in the assumption on marginal power production seems to be an appropriate approach.

The market response to a change in power demand is also affected by the separation of power markets in the EU, which is a consequence of transmission constraints, *i.e*. not enough capacity in the power lines to trade electricity freely between countries and regions. The European electricity market is hence fragmented, with few national and regional couplings. The Nordic countries trade electricity on a common market exchange, Nordpool, where the so-called system price is set based on production output and demand. However, the area price differs depending on where more electricity is produced, and even within Sweden there are now four price regions since 2011. Northern Sweden, where most of the large-scale hydro power plants are located, has a lower price than central and southern Sweden, which is more densely populated but has less available power production capacity.

The EU now has the goal of creating a single market, *i.e.* eliminating bottlenecks via interconnectors and obtaining a single areal price for power. There is a specific goal of 15% interconnection by 2030 (EC, 2014). Such coupling of markets also means that the energy mix would be the same for all of Europe, something that could influence the future environmental valuation of electricity.

6.7 Barriers to implementation of decentralised bioenergy systems

Bioenergy and biofuel production are, and have for a long time been dependent on financial policy instruments, such as investment support, feed-in tariffs, green certificates *etc*., to become economically viable (Popp *et al*., 2014; Jenssen *et al*., 2013; Hiremath *et al*., 2009; Berndes & Hansson, 2007). The systems in this thesis were for the most part not analysed from a cost perspective, but undoubtedly most of the investments required for the scenarios would be difficult or impossible for a single farmer to bear without financial support of some kind. This is true not least for biogas systems. Today there is however an investment support programme in Sweden for biogas systems, as part of the Swedish rural development programme, and the number of farm-based biogas systems is slowly increasing.

Straw boilers are, on the other hand, already quite common today for farm use and typically have a short pay-off time as they often replace use of expensive oil (Swedish Energy Agency, 2014a).

In some of the scenarios studied here, infrastructural investments were also required, *e.g.* for upgrading of biogas to LBG in a cryogenic plant. This requires both an upgrading facility and culverts to lead raw gas to the plant, an investment that requires public investment or financially very strong private investors.

The self-sufficiency systems studied in Papers II and II of this thesis produced large amounts of waste heat. If a market for this heat were available, the potential for economic viability would improve considerably. The availability of a heat market depends on the location of the farm, as heat cannot be transmitted over large distances. If the farm is located close to a village, providing district heating could be a potential second business activity for a farmer or farm cluster, or supplying heat to a greenhouse or industry. In the current market situation, with low electricity prices and over-capacity on the market, revenues from commercial production and sale of electricity from CHP production could become financially difficult. It is therefore crucial to have an off-set market for heat.

6.8 Final remarks

Biomass can be used to produce heat, cooling, electricity and transportation fuel. A key question is how available biomass resources best is used – in self-sufficiency systems (multiple outputs), large-scale power plants, decentralised CHP, district heating, advanced vehicle fuel production (such as lignocellulosic ethanol production) or other applications.

According to Popp *et al*. (2014), a high share of electricity from biomass is the easiest to achieve, a high share of heating/cooling is the most difficult (at least without DH systems), and a high share of biofuels in the transportation sector the most uncertain. This analysis may not apply everywhere (for example, fuel for DH production in Sweden is more than 40% biomass), but there is a point to it. Existing large-scale coal power plants can relatively easily be refitted to using wood pellets or forestry residues, without overly extensive technical or economic investments. Heat production systems for the residential sector in EU are on the other hand more disperse, often based on individual gas-based systems. This makes them considerably more complex to replace. Regarding the transport sector, biofuel producers in Sweden and the EU have for long been complaining about volatile policies that create uncertain market conditions - hampering innovation and willingness to invest.

Building extensive DH systems in EU based on biomass or other renewables, similar to the existing DH systems in Sweden or the decentralised systems proposed in Papers III and IV, have been proposed by the EU Roadmap for Heat 2050 (Connolly *et al*., 2013). Such systems take time, effort, political will and financial resources to build, but can be an effective measure to reduce the European dependence on natural gas.

Today, 270 000 m^3 diesel and 9 000 m^3 petrol are used in the agricultural sector in Sweden; biofuel use is limited to 4 000 m^3 RME and 100 m^3 ethanol (Swedish Energy Agency, 2013). Farm-based production of vehicle fuel would reduce dependence on a volatile petroleum market.

Producing heat, power and vehicle fuel from farm residues is possible, and can create an important carbon sink in agricultural soils if managed right - in particular in energy systems based on biogas and SRC. Building such systems would be an important step towards sustainable food production, and towards reaching the goal of limiting the mean global temperature increase to 2 ºC by 2050.

7 Main conclusions

- Both arable and dairy farms can become self-sufficient in energy by utilising on-farm residues or SRC grown on set-aside land in different technical systems
- GHG emissions from production of 1 kg ECM can be reduced by up to 46% by replacing fossil fuels with energy from on-farm biomass residues on organic dairy farms. The emissions reductions are partly due to elimination of fossil fuel use, and partly to changes in manure management and soil carbon content
- Annual GHG emissions from organic arable farms can be reduced by up to 35% by replacing fossil fuels with ley and straw. However, energy systems mainly based on straw suffer from the negative impact of crop residue removal on soil carbon and, for the organic arable farm studied here, gave only a 9% emissions reduction compared with the fossil system.
- Energy systems based on biogas from mainly ley and/or manure or systems based on SRC cultivation achieve higher GHG emissions reductions than systems involving crop residue removal.
- Local, small-scale district heating system based on agricultural and forestry residues can provide heat to a village with lower GHG emissions and production costs than a central plant with regional forestry fuel sourcing or based on fossil fuels.
- Production of heat in a local district heat system based on Salix and straw reduced GHG emissions by 97% compared with heat produced from natural gas, 78% compared with heat produced by forest residues in a central plant, and 10% compared with heat produced from locally sourced forest residues and pellets. The carbon sequestration effect of Salix contributes strongly to these results.

Acknowledgments

I wish to thank my supervisors Per-Anders Hansson, Cecilia Sundberg and Åke Nordberg, as well as the co-authors of the papers comprised by this thesis for your guidance and support throughout this work. Thank you for all the opportunities to learn and develop that I have had during this project.

Thank you to SLU's Samlade Studentkårer (SLUSS) and IVA Student Council, for taking me on board to work with intriguing issues related to academia and industry, and for a lot of good times ☺
Thank you also to my lovely SLU colleges Johanna Spångberg, Elin Röös, Olle Olsson and Marlene Areskog, for invaluable friendship during and beyond the PhD student life. Johanna – I probably wouldn't have made it without your food, music and company at our office!

Finally, thank you to my family and friends for support and encouragement on this long journey.

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