REPORT 2013:7

BIOFUELS AND LAND USE IN SWEDEN – AN OVERVIEW OF LAND-USE CHANGE EFFECTS

Report from an f3 R&D Project

Authors

IVL Swedish Environmental Research Institute (IVL)
Jonas Höglund (Project Manager)
Karin Hansen
Mathias Gustavsson
Julia Hansson

Lund University (LU)
Serina Ahlgren
Pål Börjesson

Swedish University of Agricultural Sciences (SLU)
Cecilia Sundberg
Jan-Olof Helldin
Elham Ahmadi Moghaddam

Chalmers University of Technology (Chalmers)
Maria Grahn
Martin Persson

Swedish Institute for Food and Biotechnology (SIK)
Christel Cederberg

THE SWEDISH KNOWLEDGE CENTRE FOR RENEWABLE TRANSPORTATION FUELS

FOREWORD

This project was funded by the Swedish Knowledge Centre for Renewable Transportation Fuels, f3 (Fossil Free Fuels). The f3 Centre is a nationwide Swedish centre that contributes to the development of sustainable transportation fuels by initiating research projects and syntheses of current research.

The report includes a literature review of existing knowledge related to biofuel induced land-use change effects, the identification of knowledge gaps and a synthesis of the results.

The project was conducted by IVL Swedish Research Institute (Project Manager), Lund University (LU), Swedish University of Agricultural Sciences (SLU), the Chalmers University of Technology (Chalmers) and the Swedish Institute for Food and Biotechnology (SIK).

Stockholm/Gothenburg, Sweden, January 2013

Contributing authors

IVL Swedish Environmental Research Institute (IVL)
Jonas Höglund (Project Manager)
Karin Hansen
Mathias Gustavsson
Julia Hansson

Lund University (LU)
Serina Ahlgren
Pål Börjesson

Swedish University of Agricultural Sciences (SLU)
Cecilia Sundberg
Jan-Olof Helldin
Elham Ahmadi Moghaddam

Chalmers University of Technology (Chalmers)
Maria Grahn
Martin Persson

Swedish Institute for Food and Biotechnology (SIK)
Christel Cederberg

This report shoud be cited as:

Höglund, J., Ahlgren, S., Grahn, M., Sundberg, C., *et al*, (2013) *Biofuels and Land Use in Sweden – An Overview of Land-Use Change Effects*. Report No 2013:7, f3 The Swedish Knowledge Centre for Renewable Transportation Fuels, Sweden. Available at www.f3centre.se.

EXECUTIVE SUMMARY

Supported by policies, biofuel production has been continuously increasing worldwide during recent years owing to a scientific consensus that human-induced global warming is a reality and the need to reduce import dependency of fossil fuels. However, concerns have been raised that biofuels, often advocated as the future substitute for greenhouse gas (GHG) intensive fossil fuels, may cause negative effects on the climate and the environment. When assessing GHG emissions from biofuels, the production phase of the biofuel crop is essential since this is the phase in which most of the GHG emissions occur during the life cycle of the fuel (not accounting for biogenic CO_2 from the tailpipe). Much research has been focusing on the GHG performance of biofuels, but there are also a range of other possible environmental effects of biofuel production, often linked to land use and land management.

Changes in land use can result from a wide range of anthropogenic activities including agriculture and forestry management, livestock and biofuel production. Direct effects of land-use change (LUC) range from changes of carbon stock in standing biomass to biodiversity impacts and nutrient leakage. Beside the direct effects, indirect effects can influence other uses of land through market forces across countries and continents. These indirect effects are complex to measure and observe.

This report provides an overview of a much debated issue: the connection between LUC and biofuel production and associated potential impacts on a wide range of aspects (i.e., soil chemistry, biodiversity, socioeconomics, climate change, and policy). The main purpose of the report is to give a broad overview of the literature on LUC impacts from biofuel production, not only taking into account the link between LUC and GHG, which has been addressed in many other studies.

The report first presents a review of the literature in the different scientific areas related to LUC and biofuel production. Next, knowledge gaps related to LUC is compiled and, finally, a synthesis is developed highlighting major challenges and key findings.

The synthesis identifies the following major challenges associated with biofuel-induced LUC that need to be addressed in policy-making processes:

- *Deforestation, forest management, and climate change*
  Deforestation is a major contributor to GHG emissions and can contribute to soil erosion and carbon stock changes. Other effects include albedo changes and the timing of emissions which need to be better understood.

- *Degradation of biodiversity*
  Land-use changes is one of the major threats to global biodiversity and it can be induced by most kinds of human activities. Forestry for timber or agricultural food production also induces LUC, and great care must be taken to develop sustainable biofuel production to ameliorate this impact.

- *Nutrient leakage and removal*
  Increased use of forest residues can influence the growth potential of nutrient pools, which is especially important when forestry residues are utilised for biofuel production. To avoid fertility losses in agricultural soils during biofuel production, crops with low fertilizer needs, high nutrient use efficiency and high yields should be given priority.

- *Challenges in quantifying indirect land-use change (iLUC)*
  Indirect effects on land use are extremely complex to quantify without great uncertainty.

- *Contribution to rising food prices and poverty*
  Even more challenging than quantifying iLUC is trying to measure the impact on food prices and poverty from LUC. Results show that biofuel production can have an impact on such factors.

- *Other socioeconomic aspects*
  Biofuel production can create jobs but also interfere with traditional ways of life and recreational values. To avoid negative effects, biofuel production should be developed in collaboration with the stakeholders involved: farmers, land owners, tourists, and industry.

Although biofuels can contribute to decreasing GHG emissions, there are other environmental concerns associated with biofuel production that need to be addressed: e.g., nutrient leakage, biodiversity loss, etc. The diversity of biofuel feedstock and the multiplicity of biofuel pathways lead to high uncertainty in measuring the resulting effects, especially when indirect effects are considered. Since biofuel feedstock interacts with other commodities on the market, it is a great challenge to develop relevant certification systems in order to avoid risks. The LUC caused by increasing use of biofuels can be negative to various degrees. However, the drawbacks can be mitigated through policy measures or technology developments. Examples include the cultivation of high-yielding crops, cultivation on abandoned arable land, and effective use of by-products and waste. Second generation biofuels derived from cellulosic feedstock, such as salix or poplar, also hold promises since they most likely offer higher biomass yields and increase carbon storage.

Current methods of LUC assessments typically focus on GHG emissions. However, tools and approaches to account for effects other than greenhouse gases should be adopted.

The literature review and synthesis presented in this report shows that land use on this planet is already placing high stress on ecosystems, atmosphere, soils and human life. Because of increased biofuel production, land use change is therefore at risk of aggravating these problems. To avoid these pitfalls and instead explore the opportunities that exist for beneficial land-use change, continued responsible and sensitive collaboration between industry, policy-makers, researchers and local communities is a prerequisite.

CONTENTS

LIST OF ABBREVIATIONS

LUC	Land-use change. In this report, used as a broad term to describe both direct and indirect land-use change (the sum of dLUC and iLUC). Land-use change occurs when land use is changed from one state to another (e.g. conversion of forest to agricultural land).
GHG	Greenhouse gas
dLUC	Direct land-use change
iLUC	Indirect land-use change
LCA	Life cycle assessment
CO2eq/MJ	A measure to compare emissions from various greenhouse gases expressed per megajoule (MJ) of the biofuel.
C:N-ratio	Carbon-to-nitrogen ratio. A ratio of the mass of carbon to the mass of nitrogen in a substance.
C3 plants	A carbon fixation pathway for photosynthesis. C3 plants differ from C4 plants in that they tend to thrive in areas with moderate climate (moderate sunlight, temperature and so on).
NPK (-fertilizer)	Fertilizers containing the macronutrients Nitrogen, Phosphorous and Potassium
RED	Renewable Energy Directive (2009/28/EC)
CEC	Cation exchange capacity
RF	Radiative forcing
GWP	Global warming potential
EC	European Commissio

GLOSSARY

1st generation biofuels	Biofuels derived from raw material containing sugar, starch, oil or animal fats. The majority of today's commercial biofuels are of 1st generation origin based on raw material grown on arable land.
2nd generation biofuels	Biofuels derived from lignocellulosic biomass such as wood, grasses and forest residues.
Base cations	The most prevalent cations in the soil. Base cations include ions such as calcium (Ca2+), magnesium (Mg2+) potassium (K+) and sodium (Na+).
Biochar	Charcoal generated by pyrolysis (burning) of biomass. Biochar can increase soil fertility and raise agricultural fertility.
Biofuels	Fuel derived from organic matter obtained directly from plants or from agricultural, commercial, domestic and/or industrial organic wastes.
Biomass	In this report: plant material intended as feedstock for biofuel production.
Biome	Regions with climatically and geographically similar conditions on the planet and characterized by their climate, flora and fauna, sometimes referred to as ecosystems.
Bioremediation	The use of microorganisms to remove pollutants.
Coniferous forest	Vegetation primarily composed of cone-bearing, needle-leaved, or scale-leaved evergreen trees found in regions of the world that have well-defined seasons, at least four to six frost-free months. The northern Eurasian coniferous forest is called the taiga or boreal forest.
Deciduous forest	Vegetation composed primarily of broad-leafed trees that shed all their leaves during one season. Deciduous forest is found in three middle-latitude regions with a temperate climate characterized by a winter season and year-round precipitation: Eastern North America, Western Eurasia, and Northeastern Asia. Deciduous forest also extends into more arid regions along stream banks and around bodies of water.
Land cover	There is no international agreed-on definition of land cover but usually refers to observed (bio)-physical cover of the earth's surface. According to FAO definitions (fao.faostat.org/site/377), land cover types include: forest area, inland water, agricultural area and other land,including roads and cities mountains and land with permanent ice cover and deserts.

Land use	Refers to the use of the land cover. Contrary to land cover, land use is difficult to "observe". For example, it is often difficult to decide if grasslands are used for agricultural purposes or not. There is no international agreed-on definition or classification of land use categories. According to FAO, classification is done according to the agricultural holders' concepts of use, i.e. arable land, pastures etc. Within these categories, there are sub-categories to further define use.
Nemoral climate	According to The Heinrich Walter biome classification scheme: moderate climate with freezing winters.
Phalaris	A genus of grasses.
Pollarding	Cutting shoots from high stumps.
Semi-natural coppice	Semi-natural is here defined as man-made or strongly human-influenced habitats which through a lengthy history of traditional land-use have formed characteristic biodiversity built up by the spontaneous colonisation of “wild” species.
Slash	Branches and tops (Swedish: “grot”).

1 INTRODUCTION

1.1 BACKGROUND

Replacing oil-based transportation fuels with biofuels is expected to contribute to decreased greenhouse gas emissions and reduce the dependence of imported fossil fuels. The EU has established a target of 10% renewable fuels for transport (including biofuels) by 2020 for each member state, laid down in the so-called "Renewable Energy Directive" (2009/28/EC) (EC, 2009). Many other countries also have ambitious biofuel targets, for example the US, Brazil, India and China. With growing demand for biofuels, we can expect increasing pressure on land resources and as a consequence of land-use change (LUC).

Biofuels are expected to contribute to decreased greenhouse gas emissions and reduce the dependence of imported fossil fuels. In recent years, a debate has been emerging on the sustainability issues related to biofuels, the extent to which biofuels contribute to greenhouse gas mitigation and other environmental concerns in juxtaposition to fossil fuels. In the EU, the Renewable Energy Directive adopted in 2009 (EC, 2009) strives to ensure that the biofuels used within the EU are sustainable, which has led to extensive research in the field. Research has shown that the sustainability problems associated with biofuels mostly relate to effects of land-use change due to the production of feedstock. The first widespread study that made these complications visible was a publication by Searchinger *et al* (2008) on the indirect effects of biofuels. It concluded that the effects of indirect land-use changes might exceed the positive effects of biofuels compared to fossil fuels, at least in terms of greenhouse gas emissions. Searchinger concluded that both corn and cellulosic ethanol in fact increased greenhouse gas emissions compared to gasoline. Fargione *et al*. (2008) was another influential study that evaluated different scenarios of land clearing for biofuel production and concluded that biofuel production created a carbon debt which in many cases outweighted its positive effects.

Following the Searchinger and the Fargoine studies, the emissions from land-use change has been heavily debated both among researchers and in popular media. The methodologies and assumptions made when estimating the impacts of land-use change often differ between studies which make it difficult to arrive at general conclusions. The discussions regarding the climate impacts of biofuels and land-use change were expanded to other issues related to land use, such as biodiversity loss and the effects on poverty in the world. The price spike of agricultural commodities in 2008-2009 was in large part blamed on the biofuel industry which experienced rapid growth during the period. The debate regarding these issues has been rather undifferentiated, characterizing biofuels as neither good nor bad.

The LUC concept is relevant because the resulting impact can significantly affect global climate and other environmental impacts, as well as the social and economic effects of biofuel chains. LUC is a critical component of the overall performance of a biofuel production chain as compared to using alternative fuels. The terms dLUC and iLUC distinguish between different consequences that may arise from an increased demand of bioenergy. First, new land can be taken into production to grow biofuel crops. Converting land from one state to another (e.g. from forest cover to a crop field) to grow biofuel crops is referred to as direct land-use change (dLUC). Direct land-use change, however uncertain, can be observed and measured. On the other hand, if biofuel crops were to be cultivated on already existing agricultural land, these crops might displace other crop production which may lead to land conversion elsewhere. The latter case is referred to as indirect land-use change (iLUC). iLUC effects are closely coupled with the demand and supply of agricultural commodities, which ultimately may lead to a change in market behaviour leading to changes in land use. In contrast to direct land-use change, iLUC cannot easily be observed or measured since it results from a series of consequences. Therefore, iLUC estimates tend to be more uncertain than dLUC estimates. Note that the term LUC in this report is used as a broad term to describe both dLUC and iLUC (i.e., the sum of dLUC and iLUC).

Land-use changes impact a wide range of environmental and socioeconomic aspects which have been more or less studied. Issues associated with land-use change impact a range of life support mechanisms and ecosystem services, not only greenhouse gas emissions. This report covers several interconnected fields related to land-use change, including biodiversity, soil chemistry, iLUC, climate change, socioeconomic impacts and policy development. Socioeconomic aspects include effects on cultural and recreational values as well as effects on food prices and poverty. It is known that increasing food prices mainly impact the world´s poor population, but the link between rising food prices and biofuel-induced land-use change is not obvious.

Besides impacts on socioeconomic aspects, biofuel induced land-use change has been accused of affecting biodiversity values. Also, direct and indirect impacts of biofuel production on biodiversity are highly context-specific; the effects depend on what type of habitat/land that is converted to another practise, which crops that are used, management practices applied, etc. Impacts can be much greater when areas of high biodiversity values are affected, such as rainforest.

Intensified use of land for the production of biofuels might also affect soil quality which can have effects on the soil nutrient composition, which in turn affects the future productivity of land and downstream water quality. Soil quality depends on feedstock, land management and the climate regime but practices can be adapted to minimize negative effects. Policies have been developed to try to handle the effects of land-use change; today, numerous certifications, standards and policies exist that address land-use change issues to varying degrees.

This report covers several interconnected fields related to land-use change; biodiversity, soil chemistry, iLUC, climate change, socioeconomic impacts and policy development from primarily a Swedish, but also from an international, perspective.

1.2 PURPOSE AND OBJECTIVES

The purpose of this report is to present the current state of knowledge and identify knowledge gaps related to environmental and socio-economic consequences of LUC linked to an increased production of biofuels for transportation (hereafter referred to as biofuels). The focus is mainly on impacts related to the increased production of Swedish biomass for production of biofuels, including both direct impacts in Sweden and indirect impacts in other parts of the world.

1.3 METHOD

This study is built on a literature review. Due to the interest and importance of the issues related to LUC, a large volume of research within this area has been carried out in different parts of the world. As a result, many facts are scientifically established. However, on some issues scientists have presented studies and results that diverge to a certain extent. Since the research is extensive, diverse and widespread, it is difficult to grasp what the scientific community as a whole knows and concludes on these matters. For this reason, it is important to assemble and compile research results.

The starting point is feedstock and biofuel production in Sweden, as well as Swedish biofuel use. The research on impacts of LUC on climate, biodiversity, and soil chemistry is reviewed, as well as the research on social and economic impacts, such as effects on food prices and policy development.

Today, the largest share of all biofuels used in Sweden is so called 1st generation biofuels (biofuels based on sugar, starch and vegetable oils). 2nd generation biofuels based on, for example, forest based cellulosic materials are commercially available on a small scale in Sweden (e.g., DME based on black liquor, hydrogenated oils) but are expected to grow in future importance. As a result of this expectation, the report focuses on the impacts of land-use change resulting from the increased production of lignocellulosic biofuels.

1.4 DEMARCATIONS

The impacts of land-use change are not restricted to Sweden, and the impacts of the Swedish use and production of biofuels must subsequently be studied from a global perspective (Figure 1). iLUC impacts are even more challenging to study in one country alone and the system studied must thus be expanded to a wider perspective. It has not been possible to describe all environmental effects of land-use change globally. The report, however, strives to reveal the connection between different land-use change impacts and Swedish production.

Land-use change impacts are studied through a primary production perspective throughout the report, i.e.the different aspects (biodiversity, soil chemistry etc.) are related to the production phase of the biomass intended as feedstock for biofuel production. The term “production” is, in this report, mainly used to describe the agricultural and forestry production of a biofuel, not the processing steps to convert raw materials into biofuels. Land-use change issues linked to the processing of a biofuel (processes taking place inside a production facility) are not considered. As a consequence of the abundance of research concerning land-use change and biofuels, it is neither feasible nor desirable to summarize the current state of knowledge in great detail or with high scientific resolution, but rather highlight the key results based on current research.

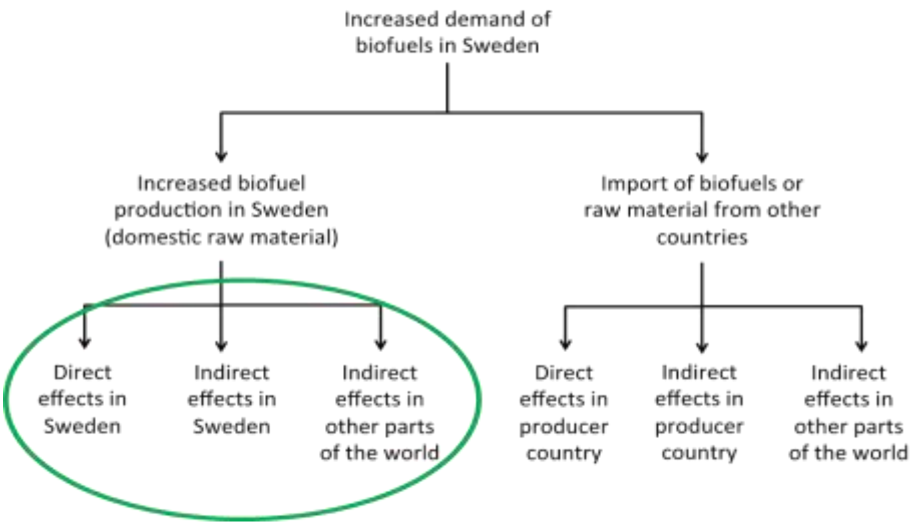

Figure 1. Increased biofuel demand in Sweden can have direct as well as indirect effects in Sweden or in other countries. The focus of this report is on impacts of increased raw material production in Sweden (left, green circle). However, production in Sweden has indirect effects globally, which are included in the report, and these are neither possible nor desirable to isolate from LUC impacts of biofuel production elsewhere in the world (right).

1.5 MAIN TARGET GROUP

The main target group of the findings in this report includes:

- Swedish policy-makers,
- Swedish industry,
- research funding organizations,
- researchers, teachers, students

Information on what is scientifically known about the impacts of LUC caused by the present and future production of biofuels is important to Swedish industry in order to obtain a solid basis for investments and strategic decisions on future biofuel production. Such information is also important to Swedish policy-makers, both for the development of national policies and for influencing the development of policies on the European and global level. Likewise, when planning future research objectives and assessing future research applications, information on what is not yet known is useful to research funding organizations. The result is also useful to researchers within this area as background material and as an aid in developing models, research questions and projects.

1.6 STRUCTURE OF THIS REPORT

The report is structured according to eight different and sometimes highly interrelated areas of knowledge linked to land-use change. The literature review is divided into topics such as greenhouse gas emissions from soils, plant nutrients and other soil chemistry aspects, biodiversity, socioeconomic aspects associated with land-use change, climate impacts of land-use change, the quantification of indirect land-use change and effects on food prices, food security, poverty and land rights, as well as policy development pertaining to land-use change. The result of the literature review is then analyzed and discussed in the Synthesis in Chapter 4.

2 LAND-USE CHANGE EFFECTS: A SWEDISH PERSPECTIVE

An increased demand for biofuels may result in a number of different land-use changes in Sweden. Land-use changes that are covered in this report include increased harvests of current vegetation, new energy crops on agricultural land and changes in soil management. More specifically, land-use changes in Sweden that may occur as a result of increased production of 2nd generation biofuels are:

- Increased harvest from forests: tops and branches of trees and trees stumps
- Harvest of residues from cropland, especially straw
- Semi-natural coppicing and pollarding in mixed forests and pastures
- Energy crops on agricultural land
  Willow
  Poplar
  Energy grasses, such as reed canary grass
- Soil management changes
  Ash recycling
  Fertilization of forests (other than ash)
  Biochar (charcoal used as a soil amendment) on cropland

Some of these land-use changes are already happening (removal of tops and branches), others have been introduced and extensively researched (willow). For some species, large research programs are in progress (stumps). Others have not been adequately researched and require further attention (poplar, reed canary grass, biochar).

In this chapter, all these land-use changes are to some extent covered under the different environmental effects investigated. Moreover, we describe abandoned and unmanaged farmland in Sweden, which may be available for bioenergy production.

2.1 CURRENT LAND USE IN SWEDEN

Sweden´s landscape is dominated by forests, lakes, wetland areas and shallow soils (Figure 2) with forests accounting for roughly half of the total land area. Almost all forest areas are used for continuous forestry of varying intensity and approximately 6 % of the forest area is protected for environmental reasons, mostly in the Western mountainous region.

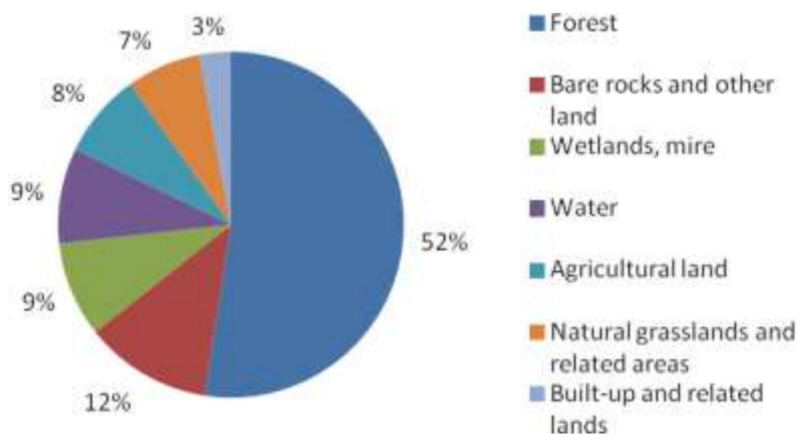

Figure 2. Percentages of different types of land use in Sweden 2008 (SCB 2008). Over 50 % of the total Swedish land area is covered by forests.

Eight vegetation zones can be distinguished throughout the country, as shown in Figure 3. Most of the land area consists of coniferous forests (the boreal zone and its sub-zones). The boreal forest is characterized by slow-growing forest with relatively nutrient-deficient soils. In Southern Sweden, there is a small region featuring nemoral climatic conditions. This area is mainly dominated by deciduous forests (KSLA 2009).

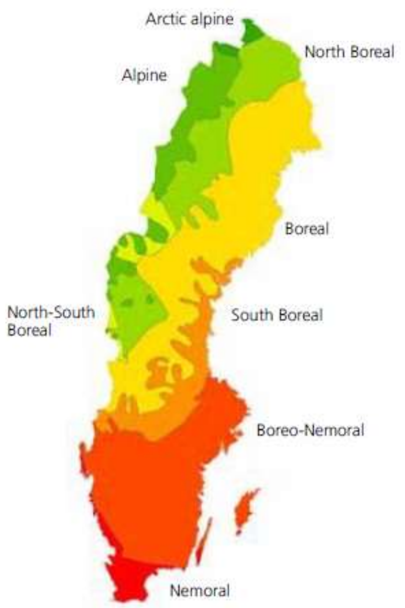

Figure 3. The eight vegetation zones in Sweden. (KSLA 2009)

2.1.1 Abandoned and unmanaged farmland in Sweden

According to 2011 agricultural statistics, total Swedish arable land area was 2,619 million ha of which 146 000 ha was fallow, i.e. the farmers voluntarily have taken 6.7% of the arable land out of production (SJV, 2012). Arable land development from 1960 until 2011 is shown in Figure 4. As seen, there has been an on-going reduction of arable land which was most accentuated during the 1960s but has been slowing down in recent years. In 2005, the EU Common Agricultural Policy (CAP) introduced new policies; subsides were given per hectare instead of per produce unit, which most likely explains why agricultural land is decreasing at a lower rate in recent years. In 2008, the CAP requirements of obligatory fallow set-aside land ceased due to increased demands for agricultural commodities on the global market (grain, soy, sugar) and strong increases in prices. During the last four years, after the boom in commodity prices, the arable land in production in Sweden has been quite stable at around 2.4 million ha.

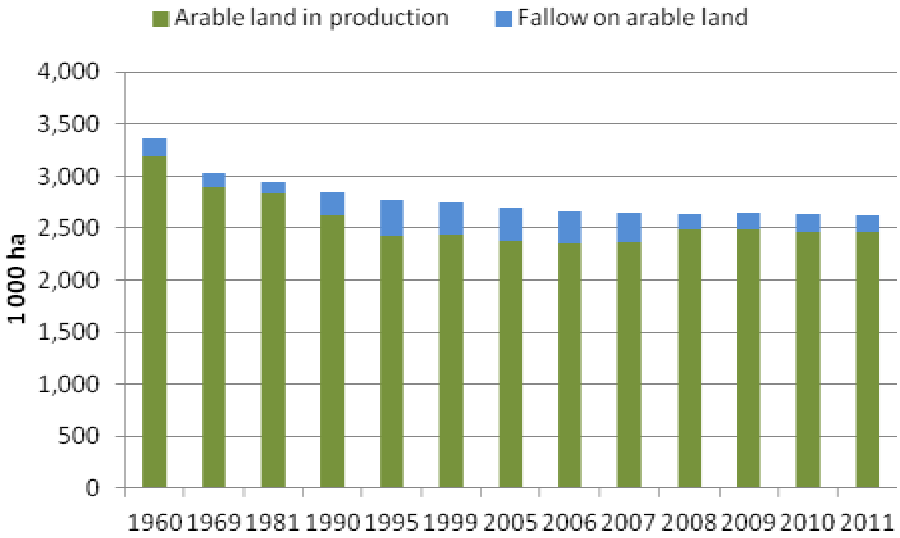

Figure 4. Area in Sweden of arable land in production vs. fallow (1 000 ha) since 1960 until 2011 (uneven x-axis) (Source; SJV 2008; SJV 2012)

The total agricultural area is the sum of arable land and grazing/meadow lands. In 2011, close to half a million ha was classified as grazing/meadow land, thus summing up the total Swedish agricultural area at 3 million ha[1] (SJV, 2012). As opposed to the situation for arable land, an environmental goal has been established to preserve grazing lands and meadows since this land use is very important for preserving biodiversity (SJV, 2008a) (for biodiversity impacts see Chapter 2.4). The goal is to preserve around half a million ha of grazing lands and meadows, an area that is decreasing by more than 10% between 2008 and 2011, from 514,000 to 447,000 ha (SJV, 2012).

Based on database information for registrations of agricultural land, the foundations for subsidies within the CAP payment program, the Board of Agriculture has quantified how much agricultural land that is out of production. The estimate was made in 2008 and only minor changes have occurred since. The analysis also includes estimates of the production potential of the unmanaged land, mostly based on field areas and the location in Sweden where the land is situated. In the SOU-report “Bioenergy from Agriculture – a Growing Resource”, it was suggested that a field size of more than 6 ha would be necessary to keep an economically viable short-willow production. Annual bioenergy crops that require more management on a yearly basis probably require larger fields while using unmanaged land for forests likely means that also fields smaller than 6 hectares

[1] Fallow arable land is included in the total agricultural area.

are viable (SJV, 2008b). The potential for unmanaged/abandoned agricultural land in Sweden is presented in

Table **1**.

Table 1. Potential for unmanaged/abandoned agricultural land in Sweden based on SJV (2008a, 2008b).

Land type	Area, hectare	Comment
Voluntary fallow on arable land	150,000	Small field areas (approximately around 1.5ha/field), a substantial share of this area is made up of field <6ha
Agricultural land with no application for subsides 1998-2006	110, 000	Here it is not possible to distinguish between arable and grazing/meadow land. Generally small field size, of the 110 000 ha, percentages of different field areas: >6ha on 12% >1ha on 64% >0.3ha on 91% Largest share of this land type is found in Norrland and lowest in Skåne and on Gotland.
Agricultural land with partly no application for subsides 2006	145, 000	Here it is not possible to distinguish between arable and grazing/meadow land. In the database, it was possible to set a field size limit of 6 ha, here it is 5 ha. Of the 145,000 ha, percentages of different field areas: >5ha on 30% >1ha on 62% >0.3ha on 81% Largest share of this land type is found in Dalarna and Norrbotten and lowest in Cental plains (around Mälardalen)
Agricultural land sorted out from the database during 1998-2006	127, 000	Here it is not possible to distinguish between arable and grazing/meadow land. These are agricultural areas sorted out by regional County Boards when it has become obvious that the land is no longer in agricultural use. There is no systematic approach to the sorting out. However, the dominant share is made up of small fields; 46% of the this area was on fields<1ha and only 20% on fields>6 ha Relatively large parts of Västra Götaland (West Sweden) are of this land type
Total	~530, 000 ha	
Agricultural land included in database but classified as forest according to Riksskogstaxeringen	-100, 000 ha	Rough estimate. A comparison between database of agricultural land and Riksskogstaxeringen is uncertain since it is not possible to directly compare maps.
LUC to urbanization	<1, 000 ha/yr	During 1996-2006 around 3, 000 ha arable land. Over en long time period. According to Riksskogstaxering, maybe 60000 ha over the last 25 years.
Total	~430, 000 ha	Less than 100,000 ha are fields larger than 6 ha.
Surplus of ley and grassland on arable land	200-300, 000 ha	Calculations of need for roughage fodder and grazing for cattle, horses and sheep matched against total area of grassland and pasture. Uncertain number.

From Table 1 it is shown that the Board of Agriculture estimates that around 430,000 ha of agricultural land appears to be unmanaged and/or abandoned ("spontan igenväxning"). Less than 100,000 ha of this land appears to be larger fields (>6ha) and thus economically viable for willow production. Also, reed canary grass could be economically less interesting on a major part of this area due to small field sizes. Thus, forest planting of poplar could be the most optimal bioenergy crop on a large share of these 430,000 ha.

One major problem associated with this scenario is the potential loss of important agricultural land areas for biodiversity. As seen in

Table **1**, for some land use categories, it is difficult to distinguish between arable land and pasture/meadow land, and there is an environmental target to preserve pasture/meadow land (SJV, 2008a). There is a potential conflict between the goals of increased biomass/biofuel production and the preservation of valuable agricultural land for biodiversity, and more knowledge is needed on this issue. Also, in Norrland (Northern Sweden), land use on arable land is dominated by long-lasting leys (permanent grassland), i.e. a land use very close to pasture/meadowland and thus important for biodiversity in the Northern parts where forest is the overall dominating use of land (Cederberg *et al*., 2007).

The Board of Agriculture calculates that there are considerable surplus agricultural areas due to "over-cultivation" of grassland on arable land, 200, 000-300, 000 ha, and this surplus land might be used for energy production. There is, however, no information on how this "over-shooting" land was calculated, nor any information on where in Sweden these areas are located and their potential field sizes. This calculation is uncertain since there are inadequate data on feed rations for cattle raised for beef. The theoretical need for land producing hay and grazing for horses may be calculated but many people keeping horses have objectives other than purely economic, for example, keeping larger areas of grassland for pasture so that the horses can freely move across larger areas. Thus, the estimate of surplus grassland might be excessive.

Based on the estimates, a large share of unmanaged and/or abandoned agricultural land appears to be situated in rural areas close to forest regions. The Board of Agriculture recently stated that in the next six-year CAP-period starting in 2014, it is necessary to direct regional support for cattle and roughage fodder production to these regions to maintain a vital rural economy and to preserve biodiversity.

2.1.2 Global potential

There are also estimates on a global scale. In a study by Cai *et al.* (2011), it was estimated that 320-702 million hectares are available for bioenergy production on abandoned and degraded cropland land and mixed crop and vegetation land. As a comparison, in a study by Campbell *et al* (2008), the global area of abandoned agriculture was estimated at 385-472 million hectares. If grassland, savannah, and shrub land with marginal productivity were considered, the number would increase to 1107-1411 million hectares (Cai *et al*., 2011). Planting second generation biofuel feedstock on all available land might according to Cai *et al.* (2011) substitute as much as 26-55% of the current liquid fossil fuel consumption worldwide.

2.2 GREENHOUSE GAS EMISSIONS FROM SOILS

Soils are essential for bioenergy production as the basis for biomass production. Soil processes affect other parts of our environment, such as air and water quality. A major issue concerns the exchange of greenhouse gases into the atmosphere. Three greenhouse gases are emitted and absorbed by soils and vegetation: carbon dioxide (CO_2) methane (CH_4) and nitrous oxide (N_2O) as part of the carbon and nitrogen cycle, respectively. Carbon dioxide from the air is captured by photosynthesis into plant biomass. Some of this carbon is transferred into the soil through the roots, when plant debris falls to the ground or is plowed down.

Various types of carbon have different turnover rates in the soil. Organic materials are decomposed and converted by microorganisms in the soil. A proportion is quickly broken down into carbon dioxide, while other fractions are converted into stable substances that remain in the soil for decades or centuries. If more carbon were added to the soil than were released, the carbon stock would increase and the CO_2 concentration in the atmosphere decrease. This is called carbon sequestration. Any carbon sequestered in the soil can be released if cultivation practices should change. The soil carbon stock is very large, so small percentage changes of the soil carbon can have significant effects on the total greenhouse gas balance of biofuels.

As land-use changes, for example through intensified harvest, changes of crops or changed soil management practices, greenhouse gas emissions are likely to change. In this chapter, we discuss the current state of knowledge of greenhouse gas emissions from soils related to land-use change in Sweden. Issues concerning agricultural and forest soils are traditionally treated differently, a convention to which we adhere. However, for bioenergy purposes, it is also relevant to discuss the interface between forests and agriculture when trees are planted on agricultural land.

2.2.1 Forest carbon dioxide balance

During forest growth, CO_2 is absorbed by trees through photosynthesis which is distributed to aboveground and belowground biomass. Some of this carbon is also added to the soil, from root exudates, root decomposition and litter that falls to the ground. Natural forests are net sequesters of carbon in our climate, but when we are interested in biomass from forests, the effects of different land-use changes on carbon balances is of interest. Some main questions are raised, which are also addressed in the Synthesis include (Egnell and Olsson, 2012):

- How does the removal of tops, branches and stumps affect the soil carbon content directly? And how does it affect the soil carbon content indirectly, by disturbing the soils and by changing the water transport of carbon from soils? The state of knowledge on this subject is described below.
- How does the extraction of more biomass (i.e. tops, branches and stumps) affect the growth of forests for the next generation or remaining trees? The main factor involves the nutrients removed by harvesting more biomass, as described in Chapter 2.3. In conclusion, forest growth seems to be unaffected by stump removal but may be negatively affected by the removal of tops and branches.
- How does the addition of ash affect the regrowth of the forest stands and thus the carbon balance (see Chapter 2.3)?
- How does more intensive forest production affect the greenhouse gas balance, if higher intensity were caused by
    i. nitrogen fertilization?

ii. adding ash to peat soils?

iii. shorter rotation periods?

Model calculations based on Swedish data performed by Egnell and Olsson (2012) show how the removal of tops, branches and stumps affect the direct carbon balance of spruce and pine in Central Sweden. If only stems were harvested, the CO_2 emissions from decomposition (mainly of tops, branches and stumps of the harvested trees) are larger than the photosynthesis of new trees, and the forests are net emitters 20 years after the harvest. After 20 years, the photosynthesis is larger than the decomposition and the forests have a net uptake of CO_2. If tops, branches and stumps were also harvested for bioenergy, the CO_2 would be emitted directly after the harvest when the biofuel was burned, instead of in slow emissions during approximately 20 years of decomposition when left on the ground. The potential climate effect of the biofuel harvest would then be the effect of emitting CO_2 earlier rather than later. The decomposition of stumps would be slower than the decomposition of tops and branches and, therefore, the potential climate effect of the stump harvest would be larger than for the harvest of tree tops and branches which decomposes quickly in the forest.

In field experiments, the combination of the direct effects of the removal of carbon in biomass and the indirect effects from the disturbance of soil are measured. Some field experiments on the effects of the removal of tops, branches and stumps have been performed in Sweden in recent years, and their results are ambiguous. Short-term experiments of a few months to a year showed minor or no effects of stump removals on CO_2 emissions at fields after harvest (Strömgren 2009). In one four-year experiment 50% higher CO_2 emissions after stump harvest were found and in another five-year experiment, total emissions were lower after stump harvests, although they were higher over the first year and a half (Grelle 2011). Consequently, there is remaining uncertainty on the effects of stump harvests on CO_2 emissions. Moreover, the removal of tops, branches and stumps has other environmental effects, which are described in section 2.4.

This chapter has been outlining how the GHG uptake and emissions from forests have been affected by land-use change in the form of increased harvests of tops, branches and stumps. The most pronounced effect is a change in the timing of CO_2 emissions, when biomass is harvested and burnt shortly after harvest, instead of decomposing slowly during many years in the forest. The timing of emissions is crucial for the overall GHG performance of a biofuel system.

When a biofuel is combusted (in a vehicle engine or a power plant), the greenhouse gases stored in the biomass are immediately released during a short time period compared to the time it would take for the greenhouse gases to slowly release during the natural decay of the biomass. The time it took for the biomass to sequester the released carbon is, however, much longer – several years or decades. What is created in the meantime between biomass combustion and the time it takes to regrow the same amount of biomass is called a “carbon debt” (representing a net increase in atmospheric greenhouse gases). The carbon debt is “payed back” when new biomass has replaced the used biomass. Depending on the time it takes to balance out the released carbon by new growth, different biofuels can therefore be fast or slow. Stumps are an example of slow biomass and salix an example of a fast biomass (Zetterberg 2011).

2.2.2 Carbon dioxide from agricultural soils and crops

In general, arable production reduces soil carbon compared to natural ecosystems, such as forests and permanent grasslands, mainly through the reduced input of carbon into soils and the regular disturbance of soils by tillage. Ploughing and other soil disturbances speed up the decomposition. The degradation of soil carbon is favored by an increased temperature and moisture content, but these factors also increase crop growth and can thus contribute to increased carbon input to soils. Biomass growth is increased by adding fertilizer, which increases carbon input to soil via the roots or the plowing of crop residues.

Annual agricultural crops bind less carbon in soils and vegetation than do perennials. Perennial vegetation binds carbon in its biomass, leading to a temporary stock of carbon, which will be greater if the plant has a long life cycle.

Consequently, changing land use from annual crops to perennials; forests, short rotation forests or energy grasses can be expected to increase the carbon stored in soils and vegetation, thus leading to reduced direct greenhouse gas emissions or even negative net emissions from agricultural land use. This development has been shown and quantified for Northern European conditions in research publications, as will be described in this chapter.

Carbon stock changes in the cultivation of energy grasses

With perennial crop soils not tilled annually and inputs of carbon through roots on the increase, the steady-state soil-carbon concentrations are generally higher than for annual crops (Hillier *et al* 2009, Anderson-Teixeira *et al* 2009). According to a review of a number of investigations, the cultivation of perennial grasses (Miscanthus, switchgrass or native mixes) increased soil organic carbon by an average of 0.1-1 tonnes per hectare per year in the top 30 cm of the soil.

In reed canary grass, high rates of carbon accumulation in belowground biomass have been measured during the first three years of cultivation in Northern Sweden (Xiong and Kätterer 2010). The total amount of carbon in the top 20 cm was 3 and 3.4 tonnes per hectare at the end of the second and third growing seasons, respectively, which is two to seven times more than for food and feed agricultural crops grown in the region. The high carbon accumulation, plus the fact that the root turnover is high (root biomass decreases substantially during the winter), is a sign of the high input of carbon to soil, which indicates the potential of reed canary grass to be a carbon sink. To arrive at complete soil balances, more detailed studies, including deeper roots, are needed.

Carbon stock changes in the cultivation of willow and poplar

Rytter (2012) calculated the carbon sequestration potential for willow and poplar plantations grown on agricultural land in Sweden. Estimations of soil carbon sequestration during a harvest cycle of 22 and 20 years for willow and poplar, respectively, were based on experimental observations of fine root production and turnover (described in Rytter (2001)) and of leaf litter observation and decay modelling. Total litter (leaf + fine root litter) production was calculated to be 2.81 tonnes of carbon per hectare per year for willow and 3.51 tonnes of carbon per hectare per year for poplar. Based on a simple model (asymptotic decay function) for the litter decomposition in forest soil (Berg and Ekbohm 1991), not taking into account the previous state of the soil, this was assumed to result in a soil carbon accumulation of 0.41 tonnes of carbon per hectare per year for willow and 0.52 tonnes of carbon per hectare per year for poplar. Fine root production is about four times larger than leaf production, but fine root decomposition is faster, which means that the resulting contribution to soil carbon is similar for leaves and fine roots.

In a long-term field experiment on a loamy sand soil in Northeastern Germany, soil carbon changes were measured in *Salix* and *Populus* plantations over a 12-year period starting in the spring before establishing the plantation compared to cultivating annual crops on the same land (Hellebrand *et al* 2010). The soil carbon content dropped during the first three years of the plantation, but then increased. The total increase over 12 years was 3.95 tonnes of carbon per hectare whereas in annual crops the soil carbon content decreased by 6.15 tonnes of carbon per hectare over the same time period (average of two different fertilization rates and non-fertilized blocks). Increases in soil carbon were significantly higher in fertilized than in non-fertilized plantations.

Carbon stock changes from straw removal

The removal of straw for bioenergy purposes means that less carbon is input into the soil with the expectation that soil carbon should decrease. However, this is not always the result found in long-term field experiments (Lafond *et al* 2009). There are indications that straw is more easily biodegraded than roots; thus, roots contribute more than straw to soil carbon when cereal is grown (Kätterer *et al* 2011). In that case, the reduction in soil carbon from straw removal would be smaller than assumed until now.

2.2.3 Nitrous oxide emissions

The emission of nitrous oxide (N_2O) from the soil is the largest source of greenhouse gas emissions in agriculture. N_2O is a potent greenhouse gas and the emission of 1 kg of N_2O yields as much impact on the climate for 100 years as the emission of about 300 kg of CO_2. About 70% of the N_2O emissions in Sweden are from agriculture (Kasimir-Klemedtsson 2009). N_2O is formed in the soil by the two microbiological processes nitrification and denitrification. By nitrification, ammonium ion reacts with oxygen to form nitrate. If there is much ammonium and little oxygen available, nitrous oxide instead of nitrate is formed. Denitrification is the conversion of nitrate to nitrogen gas that occurs in the absence of oxygen, for example in wet soils. N_2O is formed as a byproduct during nitrification. Denitrification accounts for most of the N_2O emissions from soil and nitrifications for a minor part.

The N_2O emitted from soils due to nitrification and denitrification is called direct emissions. Indirect nitrous oxide emissions occur when nitrogen compounds that have left the agricultural land in the form of nitrate or ammonia react in different ecosystems to form nitrous N_2O.

N_2O emissions from soil are difficult to measure. The emissions vary widely across fields and over time and continuous measurements are often difficult to implement. Large emissions may occur during short time periods, while remaining small for the rest of the year. The size of the emissions depends on a variety of soil and weather conditions, as well as on the cropping system.

The emission factors that IPCC recommend for calculating N_2O emissions from agriculture, which are used in most LCAs and in national climate reporting, have large uncertainties. The method is very simplistic and implies that 1 % of N added to soils as mineral fertilizers, crop residues or manure become direct N_2O emissions. Indirect emissions are estimated to be 1% of nitrate leaching and 0.75% of the losses of ammonia to the air.

N_2O is also emitted from the mineral N production when ammonia is converted to nitrate. Due to new purification technology, the N_2O emissions from mineral fertilizers sold in Sweden have decreased in recent years. The N_2O emissions are about 1.3 kg CO_2 equivalent per kg of produced mineral nitrogen. In addition, about 2.3 kg CO_2 per kg N are emitted from the fossil energy required to produce the mineral fertilizer.

Nitrous oxide from land-use change for bioenergy production

It can be expected that the use of nitrogen fertilizer for increased forest growth would lead to increased nitrous oxide emissions. However, the conclusion from research performed so far is that the climate effect of increased nitrous oxide emissions is negligible compared to the effects of the increased accumulation of carbon in biomass in fertilized forests (Nordin *et al*. 2009a, 2009b).

In a long-term field experiment in Northeastern Germany, N_2O emissions (measured four times per week for eight years) from Salix and Populus plantations were half of those from annual crops (Hellebrand *et al* 2010). As expected, N_2O emissions were higher at higher fertilization rates, both for annual crops and energy crops. The authors believe that this result is valid only for similar soils (experiment was conducted on loamy sand soils under fairly dry conditions, with a precipitation of 595 mm per year) and since higher N_2O emissions are expected from no-till systems under such conditions, results could be quite different on poorly drained and fine-textured soils,.

Nitrous oxide emissions from organic soils can be large, but variations are also large. In a four-year field experiment in Finland, N_2O emissions were measured at a site where reed canary grass was planted after peat extraction (Hyvönen *et al* 2009). N_2O emissions were much lower than IPCC default values (0.56 instead of 8.6 kg N_2O-N per ha per year. Emissions were higher than at non-planted soils at the same site but much lower than at other drained peatlands in Finland that have been planted with other agricultural crops. One reason was the high C:N ratio at the site, which has been shown to be correlated to low N_2O emissions (Klemedtsson *et al* 2005).

In conclusion, the effects of different land-use changes on nitrous oxide emissions cannot be easily predicted. N_2O emissions are large and variable. Depending on local conditions such as soil texture and moisture, different nitrous oxide emissions can result from the same changes in crops and land management. In general, N_2O emissions are higher when N fertilization rates are high and when the moisture content in soils is high.

2.2.4 Biochar and greenhouse gases

Biochar, charcoal applied to soil (also known as black carbon) is the solid product of burning biomass in the absence of oxygen, known as “pyrolysis”. It has traditionally been used in some parts of the world to improve soil productivity, for example in the Amazon. Biochar is not widely used, but there are high expectations of biochar as a method for long-term carbon sequestration, as well as soil improvement. Intensive research activity is in progress on biochar internationally.

Biochar is a fine-grained residue high in carbon that can be produced from biomass such as woody material, straw, grasses, bark, sorghum, corn and sewage wastes. Biochar is a persistent material suitable for carbon sequestration that benefits the environment in terms of the mitigation of carbon dioxide. Biochar application in soil affects the physical, chemical and biological characteristics of soil along with soil fertility. Wolf (2008) and Alvum-Toll *et al*., (2011) showed that the potential global production of biochar from crop residues could be 1 million tonnes carbon/year.

Since it is long-lived and resistant to chemical processes, such as oxidation to carbon dioxide or reduction to methane (Woolf, 2008), biochar is a suitable form for sequestering carbon. Since biochar is resistant against chemical and microbial degradation, its application to soils is considered to be a CO_2 sink by transferring fast-cycling carbon from the atmosphere–biosphere system into much slower cycling carbon forms that persist for millennia (Foereid *et al*., 2011). Biochar mixed into soil is resistant to further degradation with a half-life typically around 5,000 years (Hylander *et al*., 2010). By biochar application, photosynthetically fixed carbon is added to soil, which contributes to longer carbon storage, thus mitigating increasing atmospheric CO_2 concentrations from land-use change (Lal, 2008).

Humic substances contain both carbon and nitrogen, with a fairly stable C:N ratio. Therefore, soils that are net sinks for carbon act as sinks for nitrogen as well. Similarly, soils losing carbon also lose nitrogen oxides which are destructive to the atmosphere. An experiment by Almers (2009) on the biochar effect on nitrous oxide emissions in Swedish sandy soils showed that nitrous oxide emissions were generally lower from biochar amended columns. The effect on nitrous oxide emissions seems to be partly mitigated by the increase in pH based on the biochar addition. Among other factors, nitrous oxide emissions were also negatively correlated with the specific area of the biochar (Almers, 2009).

Bioenergy and greenhouse gases from organic soils

The growth of bioenergy on organic soils is being discouraged, for example by the EU Renewable Energy Directive Sustainability Criteria based on the reasoning that GHG emissions are much higher from the production of biomass on organic soils than on mineral soils. However, from a land use perspective, the organic soils so prevalent in Sweden are major net emitters of greenhouse gases (CO_2, N_2O and CH_4). From a land use GHG emission perspective, it is important to find the best uses for these soils, i.e. practices that reduce greenhouse gas emissions. Whether bioenergy production in some form might be a good management option for organic soils, whether forested or agricultural, remains to be seen. A study in Finland investigated energy crop production on organic soils and found lower greenhouse gas emissions by using energy crops than using annual food crops (Hyvönen *et al* 2009, Shurpali *et al* 2009), but more research covering various crops and organic soils with variable peat characteristics and hydrological conditions are needed (Maljanen *et al* 2010).

2.2.5 Knowledge gaps

There is a need for deeper knowledge regarding greenhouse gas balances based on measurements of the three main greenhouse gases at an increased intensity of harvesting and production. A knowledge base for systems analyses is desirable in order to facilitate priorities between different biomass sources from a climate perspective. Egnell and Olsson call for new knowledge on nitrous oxide emissions from forest fertilization and stump harvest. Since the research performed produces contradictory results (see above), the indirect effects of stump harvest on CO_2 emissions need further attention. Key questions include how forest growth is affected by land use such as harvesting branches, tops and stumps, as well as nutrient compensation by fertilization with ash, as this is important for the carbon storage in the next forest generation. Moreover, additional research is needed on the water transport of carbon from forest areas in a landscape perspective.

Forestry with clear-cutting followed by replanting has been completely dominant for a long time in Sweden. Consequently, not much research has been performed on forestry with continuous tree cover. The knowledge gaps are large and have implications for bioenergy harvest and climate change. Some researchers claim that forestry with continuous tree cover can yield forest production with less greenhouse gas emissions (Lindroth *et al.*, 2012). Research on continuous forestry in Sweden is being performed in the Future Forests at SLU Research Program.

As land-use changes from annual food crops to perennial energy crops, deeper knowledge on the potential for storing carbon in agricultural soils is needed. For a full understanding of the differences in the soil carbon balances between crops and their causes, further studies, including long-term experiments, are needed (Anderson-Teixeira, 2009, p. 81). For better knowledge of the soil carbon sequestration potential of reed canary grass, studies of roots and carbon turnover in soils, including deep soil horizons, are needed (Xiong and Kätterer 2010).

Nitrous oxide emissions are the most uncertain source of GHG emissions from biomass production (Röös *et al* 2011). Research is called for on how emissions are affected by different types of cropping systems (food as well as energy crops) and management activities. There is a need for more advanced models that take into account factors such as climate, crops, soil quality, fertilizer types, etc.

2.3 PLANT NUTRIENTS AND OTHER SOIL CHEMISTRY ASPECTS

2.3.1 Forest soils and plant nutrients

Energy crops are cultivated with the aim of reducing the dependence on fossil fuel resources and the emission of greenhouse gases, but the production of energy crops should avoid other environmental problems, many of which are linked to fertilizer application. Nitrogen (N) and phosphorous (P) are essential elements in achieving optimum crop yield, including bioenergy crops where high biomass yield is a major aim. However, elements applied as fertilizers are not completely absorbed by plants and, therefore, some elements are released into the environment (water and air). Also, plant residues after harvest decompose and nutrients are released. This increased pressure for biomass production influences the management of forest land. Such intensification is not a specific land-use change, but rather a change in land use management. Besides already used rest products and damaged timber, the market is interested in using branches, tops and needles, which traditionally have been left to decay in the forest, as well as smaller stems from cleanings or initial thinnings and stumps after clear-cutting. Stump harvesting is of increasing interest due to its potential as a fuel. A change in the intensity of the forest production leading to increased use of different parts of the forest biomass affects future growth potentials and nutrient pools in forest biomass and soils.

When the forest is used intensively for biomass production, the control of the nutrient balance becomes important to avoid a loss of growth potential, as well as the acidification and eutrophication in forest soils and surrounding surface waters. In the long-term perspective, the wood production capacity in forests is influenced by the inputs and outputs of nutrients. Inputs of nutrients are supplied through wet and dry deposition, the weathering of soil minerals, nitrogen-fixation and fertilization or ash recycling. Outputs from the forest ecosystem take place through harvesting, leaching and erosion (Figure 5). If the nutrient budget (input minus output) is balanced, nutrient stores are assumed to be neither increased nor decreased over the time period addressed. A positive budget indicates that an element is accumulated in the system while a negative budget suggests that export exceeds import and as such, the nutrient studied is depleted from the system. The possible consequence of a decreased nutrient budget is a decreased wood production capacity. A negative nutrient balance may be interpreted as a warning of a long-term negative effect.

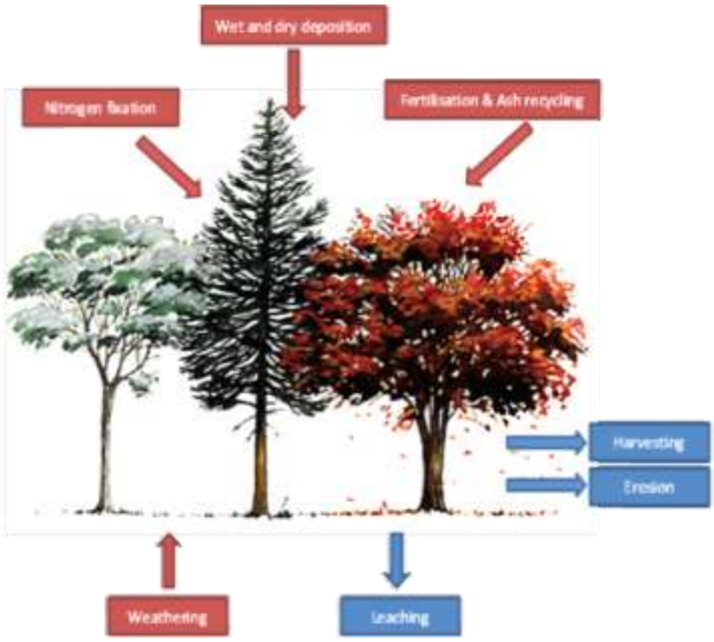

Figure 5. Possible inputs (red) and outputs (blue) of nutrients to the forest ecosystem.

Since many forest soils in Sweden are short of available N, nitrogen (N) has often been in focus in nutrient balances. Nitrogen related effects have been demonstrated, but also shortages of phosphorous (P), calcium (Ca) and potassium (K) may become a problem in areas with significant N deposition as in Southern Sweden. A deficit of N may in some areas be alleviated by high atmospheric depositions of N, while atmospheric depositions of sea salt might provide a substantial nutrient input of especially magnesium (Mg), but also K and Ca to the system on sites located in the vicinity of the sea (Kreutzer, 1995). Increased harvests of biomass from forests for e.g. biofuel production, might lead to the increased removal of nutrients, hence the importance of essential nutrient balances.

2.3.2 Acidification and eutrophication – change caused by intensified forest harvesting

When the acid production rate exceeds the acid neutralisation rate, soils become acid. Since the ice age, a slow natural acidification has taken place in forest soils through the storage of organic material. However, most acidification of soils has anthropogenic causes. From the 1950's, acidifying air pollution (S and N) has caused a significant acidification of forest soils and surface waters in Sweden. Two of the most important acidifying processes in nearly all forest soils include i) biomass growth and subsequent harvesting and ii) nitrification and subsequent leaching of nitrate and base cations. The intensity of the acidity can be measured by the pH.

Intensified forest production has contributed to enhanced soil acidity and as such, an increased acidification as an undesirable effect of the nutrient export (Karltun *et al*., 2008). When trees grow, they absorb nutrients (positive ions like ammonium (NH_4), Mg, Ca and K) from the soil and alongside hydrogen (H), ions are released causing the soil to acidify. Where no forestry is performed, the base cations are brought back into the soil when trees die and decay, thereby circulating the nutrients back into the soil to be used for further growth. However, if biomass along with base cations is removed from the forest, the acidification becomes permanent, leading to lower base saturation. Soil acidification is naturally counteracted by the weathering and deposition of base cations (Van Breemen *et al*., 1983; Raulund-Rasmussen *et al*., 1998; Akselsson & Westling, 2005).

In high deposition areas, air pollution is considered the most important source of acidity, whereas biomass harvesting may be more important when performed at an intensive level (Adams *et al*., 2000; Watmough & Dillon, 2003; Duncker *et al*., 2010), especially in low deposition areas. Clear-cutting may increase soil acidity and the loss of base cations, as shown by Likens *et al*. (1970), Mann *et al*. (1988), Keenan and Kimmins (1993) and Simard *et al*. (2001).

There are strong links between the acidification of forest soils and effects on surface waters. The percentage of lakes classified as acidified in Sweden has been declining over the last 20-30 years and will continue to decline slightly; however, a fairly large percentage of lakes are still consistently acidified. The recovery from acidification (both in forest soils and in surface waters) is a slow process and a full recovery is not only dependent on a decreased acid deposition, but also on the outtake of biomass from forest ecosystems.

Furthermore, forestry plays an important role in the eutrophication of surface waters. A significant amount of N is leaching from forest ecosystems, but by far the largest amount of N is the natural leaching of organic N (Löfgren & Westling, 2002). The largest part of leaching of inorganic N in the form of nitrate mainly takes place after land-use changes, such as forest clear-cutting, where the uptake of N decreases considerably at the same time as nitrification is favored (Akselsson *et al*., 2004) causing nitrate leaching. Only the inorganic N leaching is acidifying. However, the outtake of N rich woody biomass, which has been growing in N rich environments, may even have a positive effect on the N balance which counterbalances the risk of nitrate leaching. In this way, the outtake of biomass from the forest might have a positive effect on the N-balance.

Forest harvesting practice

The forest stand may be harvested for various products during a forest rotation. The first harvests are cleanings or initial thinnings. The number of thinnings through a rotation may vary from none to several (e.g. up to every second year), depending on the growth and thinning strategy for the stand. Commercial stem parts (stem-wood-harvesting) or sometimes whole trees (whole-tree-harvesting) are being removed, perhaps after seasoning causing the needles to be shed in the stand. The final forestry operation in a rotation is final felling and harvest. The same management regime is also applied to most forestry activities related to biomass for energy removal.

Although branches, tops and needles only account for a small proportion of the total weight of the tree biomass, they have a much higher nutrient concentration than stem wood (Pitman, 2006). A significant amount of nutrients (N, P, Ca, Mg, K) is thus exported from the ecosystem in harvesting (Glatzel, 1990; Augusto *et al*., 2002). On the other hand, stumps and coarse roots have been found to possess roughly the same content of nutrients as stem wood depending on how many nutrient rich fine roots that follow the harvest (Hellsten *et al*., 2009; Egnell *et al*., 2007). Compared to harvesting of stems alone, whole-tree-harvesting, including the harvesting of stumps and roots, theoretically may cause the average removal of nutrients per year over one rotation to increase more than six times for N and P (Nihlgård, 1972; Ballard, 2000) and twice for K (Goulding & Stevens, 1988). A consequence of nutrient removal in harvesting is that reserves may become depleted unless the mineral weathering is rapid, the atmospheric deposition of base cations is high, or a fertilisation/ash recycling is performed. When forest land is used more intensively for the production of bioenergy, the higher intensity of harvesting and the intensified export of nutrients from the forest are especially important.

Another important disadvantage of an intensive biomass outtake is that it is followed by more frequent off-road driving in the forest. Off-road driving with heavy machinery might cause soil compaction (Eliasson, 2005). The root environment will deteriorate and roots will have difficulties of extending during dry summers and wet winters because of a lack of oxygen (Froehlich *et al*., 1986; Wert and Thomas, 1981; Cullen *et al*., 1991; Wang *et al*., 2005; Eisenbies *et al*., 2005) which may influence production rates. The compaction of soils with a large content of organic material is furthermore expected to increase the concentrations of mercury in surface waters (Garcia & Carignan, 2000; Munthe & Hultberg, 2004), leading to high mercury contents in fish that harm birds and people who eat them.

Nutrient removals and growth

Studies have shown a negative effect of intensive biomass harvesting on productivity in the short-term (e.g. Jacobson *et al*., 2000; Smith *et al*., 2000; Scott & Dean, 2006; Walmsley et sl., 2009; Helmisaari *et al*., 2011). For example, Jacobson *et al* (2000) showed that whole-tree-harvesting during the first thinning generally had a negative impact on growth ten years after harvesting, even if there was a large variation among sites. It has been reported that the retention of harvest residues in the forest stand improved tree growth in the short-term (Chen & Xu, 2005; Mendham *et al*., 2003). Other studies, however, showed no significant short-term negative production effect of intensive biomass harvesting in thinnings (e.g. Powers *et al*., 2005; Nord-Larsen, 2002), indicating that soil weathering and atmospheric deposition were able to supply the stand with sufficient amounts of nutrients, at least in the short-term. Data from 58 published studies around the world where branches, tops, needles and stems were removed and compared to stem-wood-harvesting showed that approximately 80% of the studies revealed no effect on tree growth, 13% observed decreased growth and 2% showed increased growth (de Jong *et al*., 2012). Egnell (2011) investigated how long the effect of negative growth would withhold and found that growth would be slower (10-15 years) for stands (on rather nutrient poor soils) being intensively used and argued that the period probably would be shorter on more nutrient rich soils. Long-term studies (20-50 years) of impacts on growth are not yet available. The outtake of stumps is not assumed to decrease productivity (Kardell, 2010; de Jong *et al*., 2012), but only a few studies exist.

Nutrient removals and nutrient cycling

The nutrient balance approach has been used in several studies to quantify the effect of intensified biomass utilization on the ecosystem nutrient status (Boyle *et al*., 1973; White, 1974; Boyle, 1976; Olsson, 1996; Nilsson *et al*., 1998; Møller, 2000; Joki-Heiskala *et al*., 2003; Akselsson & Westling, 2005; Akselsson *et al*., 2008). Studies have shown that biomass export is related to a decrease in soil storage of nutrients, often in Ca (Likens *et al*., 1998; Huntington & Ryan, 1990), as well as a loss of exchangeable Ca measured at soil water concentrations at 50 cm depth as a result of whole-tree-harvesting (Zetterberg & Olsson, 2011). There are no clear indications of the effect on nitrate leaching after an outtake of biomass (stems, branches and leaves/needles) since studies show both nitrate leaching and its absence after whole-tree-harvesting compared to stem-wood-harvesting (Gundersen *et al*., 2006, de Jong *et al*., 2012). In a study by Zetterberg & Olsson (2011), whole-tree-harvesting resulted in lower soil water pH and acid-neutralizing capacity. For a lengthy period, the effect of biomass outtake on the acidification of forest soils and surface waters has been well established (van Breemen *et al*., 1983, Nordén, 1994). A study by Brandtberg & Olsson (described in de Jong *et al*., 2012) compared pH and exchangeable base cations for 15 and 25 years after biomass (stem-wood-harvesting and whole-tree-harvesting) had been removed. It was observed that the pH in the two plots were again practically similar after 15 years while the amount of exchangeable cations were still lower in the whole-tree-harvesting plot after 25 years.

Watt *et al*. (2005) and Powers *et al*. (2005) found that concentrations of total soil C, N and P were reduced after harvest of all aboveground living vegetation, and other researchers have likewise found that whole-tree-harvesting significantly reduces soil concentrations of N (Merino & Edeso, 1999) and P (Mroz *et al*., 1985; Tuttle *et al*., 1985; Sanchez *et al*., 2006). Reductions in concentrations of these elements may have been caused through the removal of the humus layer and topsoil in disturbed plots during harvesting, thus reducing the potential availability of nutrients. In other studies, harvesting had no significant effect on soil N (and carbon) (Johnson & Curtis, 2001; Sanchez *et al*., 2006). On the average, residue removal caused a 6% reduction in the upper soil N pools, whereas leaving residues on the site caused an 18% increase in soil mineral N content (Chen & Xu, 2005; Mendham *et al*., 2003). The retention of residues after harvest might result in higher quantities of soil exchangeable K, Ca and Mg as seen by Mendham *et al*. (2003). Soil total carbon and N contents in the topsoil have also been reported to be considerably higher when residues are retained on the site compared to when they are removed (Chen & Xu, 2005), while Olsson *et al*. (1996) found no such impact. The positive effect on soil N of leaving residues on site seems to be restricted to coniferous species. Several studies have shown that residues had little or no effect on soil N in hardwood or mixed forests (Hendrickson *et al*., 1989, Mattson & Swank, 1989, Knoepp & Swank, 1997, Johnson & Todd, 1998, Johnson & Curtis, 2001). Ring *et al*. (2001) and Högbom *et al*. (2008) observed a positive relationship between the retention of residues and concentrations of nitrate in soil solutions. Also, other nutrient concentrations like Ca, Mg, K and P have been observed to increase in soil solutions when residues have been retained on the site compared to when they have been removed (Wall, 2008).

Stump removal results in a relatively significant land-use change. When stumps are removed, the soil is disturbed rather vigorously and the effect will be a mixing of the soil which probably is comparable to heavy soil preparation methods. An environmental assessment of multiple effects of stump harvesting in Sweden has been compiled (Egnell *et al*., 2007). The authors conclude that the amount of inorganic N probably increases after stump harvesting and that there is an increased risk of the loss of nutrients through leaching, erosion and sediment transport to adjacent surface waters. Staaf & Olsson (1994) observed increased concentrations of nitrate in soil solutions for several years after stump removals. Zabrowski *et al*. (2008) analyzed a long-term effect (22-29 years) of Douglas-fir stump removal in the U.S. where a decrease in the N and C mineral soil content was observed (20-24%). Preliminary results of a Swedish experiment with an outtake of branches, needles and stumps (by Strömgren *et al*., described in de Jong *et al*., 2012) show significantly lower N content in the upper forest floor after 25 years of stump removal, whereas no change was registered in the mineral soil.

There is a substantial difference in the size of nutrient removals, depending on the size and age (Ranger *et al*., 1995) of tree species and the density of the trees at the time of cutting (Cole & Rapp, 1980; Perala & Alban, 1982; Glatzel, 1990; Augusto *et al*., 2000), site productivity, harvesting intensity, and nutrient concentration level in the biomass (Stupak *et al*., 2007a). The outtake is highest in spruce forests growing on nutrient rich soils (Hellsten *et al*., 2010). Initial planting density and the applied thinning regime seem to be of less importance for the size of the average biomass and nutrient removal.

2.3.3 Compensation through fertilization and wood ash recycling

The higher outtake of biomass from forests may be compensated by fertilization as well as wood ash recycling.

Fertilization

Nitrogen (N) is the most commonly applied nutrient when compensating forest land for a high outtake of biomass. Phosphorous and K are often added as well. The aims of fertilization are: i) to gain a positive growth response, ii) to improve or sustain good soil quality in the long-term, e.g. following intensive harvesting**.**

The compensation of nutrients can be either performed according to normal forest fertilization advice and practice (SKSFS 2007) or it can be performed more intensively on a repetitive basis (every two to three years) in younger forests according to the need of the trees established using analyses of foliage. Such “need fertilization” (“behovsanpassad gödsling” in Swedish) is supposed to yield high growth increases per kg fertilizer at the point needed by the trees. This method has recently been suggested in Sweden and only a few experiments have been performed on the environmental effects of such a fertilization regime. An analysis of the method (Nordin *et al*., 2009a) showed a great potential for a larger production of biomass.

Growth responses following fertilization with N alone or in combination with P and K have been intensively studied and reviewed in the Nordic countries (Nohrstedt, 2001; Saarsalmi *et al*, 2001; Vejre *et al*., 2001). Most commonly, growth responds positively to fertilization with N on mineral soils (Jacobson *et al*., 2000), but several experiments also showed no response. Only a few experiments showed a positive growth response upon fertilization with P alone or with P and K together, whereas several experiments showed a synergistic response when N and other elements were added together (e.g. Simcock *et al*., 2006). Fertilization with P and K has, however, shown substantial positive growth responses on peat soils. Such responses may indicate that growth is limited by the nutrient shortage in the soil. Furthermore, continued fertilization will normally lead to an increase in the soil nutrient pool and thus improve soil quality and the capacity of the soil to sustain high production. However, in areas where the atmospheric deposition of N exceeds the export of N due to harvesting, it may be hypothesized that the growth is no more N limited, as typical of boreal and temperate forests, but rather limited by other nutrients, such as P (Gundersen *et al*., 2006). Ongoing N fertilization with substantial doses has also indicated that growth becomes limited by other elements (Pettersson & Högbom, 2004).

If N were added in amounts higher than those needed to support optimal growth, it would be leached as nitrate and thus result in soil acidification, depending on the weathering capacity of the soil (Gundersen *et al*., 2006). It is, therefore, important for the fertilizer to possess a balanced nutrient composition and that it is added in balanced amounts. Experiments in Sweden with repetitive fertilization of young forest showed that nitrate was always leached in varying amounts (Nordin *et al*., 2009a; Nordin *et al*., 2009b; Bergh *et al*., 2010) and that the leaching increased exponentially as the N dose increased (Nordin *et al*., 2009b). The authors estimated that nitrate leaching from forests to coastal waters assuming an intensive use of biomass and need fertilization at 5% of the forest land, would increase nitrate leaching by 33%.

Wood ash recycling

Returning wood ash after incineration of the forest fuel has become relevant when the use of biomass for bioenergy intensifies. The principal aims of recycling the wood ash to the forest are to: i) avoid depletion of essential soil nutrients to sustain the production and to ii) sustain soil pH and base saturation in order to reduce harmful acidification of forest soils and adjacent waters (Aronsson & Ekelund, 2004).

The major components of wood ash are Ca, K, Mg, silicon (Si), Al, iron (Fe) and P, as well as trace elements, some of which are toxic (Nilsson & Timm, 1983; Steenari *et al*., 1999; Holmroos, 1993; Eriksson & Börjesson, 1991; Kofman, 1987; Booth *et al*., 1990, Karltun *et al*., 2008). Ash is generally low in N and S because these elements are volatilized during combustion. Due to varying soil mobility of toxic elements like cadmium (Cd) and caesium (Cs), caution must be exercised when wood ash is applied to forests (Karltun *et al*., 2008).

In areas where N is the growth limiting nutrient, the addition of other nutrients to wood ash is not expected to increase growth on mineral soils; in these cases, the addition of N is needed. After the addition of wood ash, Sikström *et al*. (2009) showed decreased growth on nutrient poor sites, increased growth on nutrient rich sites and no change in growth on medium sites. The addition of wood ash to forest stands on nutrient rich peat soils has shown a significantly positive effect on tree growth (Ferm *et al*., 1992) and improved conditions for the regeneration of stands (Huikari, 1951 cited in Röser *et al*., 2008; Lukkala, 1955; Lukkala, 1951). Peat soils deficient in K and P but with decent N status show the highest increase in tree growth (Silfverberg & Moilanen, 2000) after the wood ash addition, while the tree growth on peat soils low in N (<1%) remains low (Silfverberg & Huikari, 1985 cited in Röser *et al*., 2008; Silfverberg & Issakainen, 1987 cited in Röser *et al*., 2008). In conclusion, the addition of wood ash is not always beneficial to growth since growth mostly increases on soils with good N status.

When wood ash is applied to forests, soil pH rises in the upper part of the soil profile. Untreated ash yields the largest and most rapid pH increases and the higher the dose, the higher the increase in pH. The effect of wood ash on pH seems to be relatively long lasting. Ash doses around 3-5 tons per hectare have been shown to elevate pH 1 to 2 pH units in the humus layer 10-19 years after application (Mälkönen, 1996 cited in Röser *et al*., 2008; Bramryd & Fransman, 1995; Saarsalmi *et al*., 2001). The transport of ash components down through the profile is, however, slow and the effects deeper into the profile are found to be minor, usually only occurring considerably later than (>10 yrs) the ash application (Bramryd & Fransman, 1995; Saarsalmi *et al*., 2001). Hence, an increase in the pH of mineral soils is not usually found (Ring *et al*., 1999; Arvidsson, 2001; Fransman & Nihlgård, 1995) except when high doses (>10 t ha^{-1}) have been applied (Kahl *et al*., 1996).

Elevated concentrations of K, Ca and Mg can be found in the soil solution in deeper soil horizons shortly after the ash application (Westling *et al*., 2004; Norström, 2010) but the leaching of Ca and Mg is slower (Rumpf *et al*., 2001; Arvidsson, 2001). In a recent experiment where 8 t ha^{-1} wood ash was applied to a Norwegian spruce forest, pH increased from 3.2 to 4.8 base saturation increased from 30% to 86% and the BC/Al ratio increased from 1.5 to 5.5 (Brunner *et al*., 2004).

2.3.4 Agricultural soils and plant nutrients

Inputs of N and P fertilizers can be an indicator of potential environmental consequences of bioenergy cropping, such as leakage of nutrients to ground and surface waters along with atmospheric emissions (Sanderson and Adler, 2008). It is primarily important to understand the fate of an elemental application into cropping systems and the ensuing environmental effects. The fate of land-applied N and P from different sources (fertilizer application, legume residues, municipal animal or industrial wastes etc.) is as follows (Crutzen *et al*., 2008, Robertson *et al*., 2011):

Nutrient recovery rates vary for different crops and soils. Nutrient use efficiency (NUE) of crops is an important factor in determining nutrient recovery in crops. NUE is usually defined as yield per unit of input fertilizer and describes the efficiency of turning water, nutrients and CO_2 into biomass. Most energy crops for second generation biofuels are selected from perennial crops which generally have a higher NUE than common agricultural crops. Willow as a woody crop, and reed canary grass as perennial C3 plants, possess high NUE values. Low nitrogen requirement and high nitrogen use efficiency are beneficial as the impacts of fertilizer application can be minimized (Byrt *et al*., 2011, Sanderson and Adler, 2008). Studies by Boehmela (2008) show that willow, as a perennial energy crop, compared to other energy crops such as maize, has the potential for combining high yields with low inputs.

Losses as gas

Partially, nitrogen in the plant root zone is lost to the atmosphere through denitrification. Denitrification is a process of nitrate breakdown into simple nitrogen (N_2) and oxygen (O_2), gases that return to the atmosphere. In soil with high organic content and shallow groundwater, much of the nitrogen is lost in gaseous forms rather than as nitrate. This loss is less in well-aereated and low organic content soil. Due to lengthy rotation cropping systems and extendend root systems, this loss is low in bioenergy crop systems. Nitrogen in soils is also released in the form of a greenhouse gas, nitrous oxide.

Nitrogen from fertilizer application may leach to streams and water bodies through surface or subsurface drainage. This loss is considerable during heavy rains immediately after fertilizer application. Nutrient elements in the surface runoff is low if the fertilizer is mixed with the soil. The main form of nutrient leaching is as subsurface drainage in the form of nitrate. A major impact of nitrogen loss to water sources is eutrophication. Studies by Makeschin (1994) showed reduced nitrate leaching of approximately 50% in an unfertilized willow plantation in comparison to arable land for a period of three years. Willow has the capacity for high nitrogen uptake and several studies show the potential of removing nitrates from wastewater (Aronsson *et al*., 2000, 2001).

Biochar and nutrients

Biochar application affects soil nutrient cycles and nutrient leaching. Biochar improves soil chemical properties due to its high cation exchange capacity. High cation exchange capacity and high porosity of biochar increase nutrient availability and water holding capacity, while decreasing nitrogen leaching. Biochar holds nutrient elements in a plant-available form but also absorbs organic compounds and, therefore, has the potential for use in the treatment of wastewater and water polluted by toxic by-products (Sohi *et al* 2010).

The two main functions of biochar in relation to crop yield are 1) an increase in the nutrient availability of soil and 2) the stimulation of soil microbial population which leads to an increase in nutrient cycling. Biochar reduces soil acidity and decreases the need of liming. In the case of biochar use, the need for fertilizer is decreased due to its high nutrient retention.

2.3.5 Agricultural soils and toxicity

For high biomass production in agriculture, fertilizers, pesticides and herbicides are used. High applications of chemicals result in toxicity of soil and water sources. Most perennial bioenergy crops, such as *Salix* and reed canary grass, cause less toxicity in the environment due to their long-rotation cropping system and dense roots.

Studies propose *Salix* for bioenergy purposes to be planted on polluted soils for the treatment of such soils (bioremediation) (Rowe *et al*., 2009). Apart from the possibility of discovering toxic chemicals from disposed material in landfills, the main potentially polluting compounds are ammonia which is toxic to fish. However, studies in Sweden show that high salt contents, especially sodium, limit *Salix* growth (Britt *et al*., 2002).

Biochar is able to decrease the leakage of chemicals, such as pesticides, to ground and surface waters. Studies show that biochar has the capacity to speed up the mineralization of other soil organic matter. Biochar also absorbs organic compounds and, therefore, has the potential for being used in the treatment of wastewater and water polluted by toxic by-products (Sohi *et al*., 2010). A study of biochar effects on adsorption or detoxification of toxins is in progress (Elmer, *et al*., 2011). Since it reduces the availability of heavy metals in plants (Winsley, 2007), biochar has the potential for being used in the treatment of polluted soils.

Bioenergy has the potential of contributing to environmental restoration and increased productivity from contaminated land, so-called bownfields – previous industrial sites that would need costly restoration before being used for construction or agriculture (Bardos *et al* 2011). In locations where there is no economic impetus for restoration, the combination of biomass cultivation and soil rehabilitation might provide the leverage to bring economically marginal lands back into use. Potential benefits of growing biomass on contaminated marginal land include:

- reducing the climate impact of land remediation,
- generating renewable energy,
- providing broader community benefits such as landscape management,
- potentially wider environmental benefits, for example soil functionality and biodiversity,
- economic benefits, including revenue generation or offsetting remediation costs.

2.3.6 Knowledge gaps

Regarding plant nutrients and forest soils, the empirical short-term knowledge about the effect of the outtake of biomass in all forms, as well as the effect of wood ash recycling on production as well as soil and water quality, is mostly available; however, more general knowledge based on long-term experiments is needed for the understanding of the long-term effects of biomass outtake, as well as for the formulation of future recommendations. Such long-term data is even needed as data input to models and to improve the connection between models and experiments. A joint analysis of all experimental material will allow for a generation of additional answers on the possible effects on production and the environment. More details on the knowledge gaps related to long-term biomass outtake are described in de Jong *et al*. (2012).

The outtake of stumps and “need fertilisation” are new practices within forest management where there is a clear need for fresh knowledge of environmental effects. Even Nordin *et al*. (2009a) highlight knowledge gaps where more research is required on “need fertilisation”.

Continued work is needed to transform present experimental knowledge on wood ash recycling into practical guidelines in order to optimize the dose leading to positive growth effects, while at the same time avoiding unwanted environmental effects. Additional knowledge of the possible effects of the combinations of the recycling of wood ash along with N fertilization would be desirable, especially in Northern Sweden.

In general, there is a need for information on the possible effects of climate change in future scenarios of forest management where biomass is intensified.

The research on biochar is at an early stage and so far, there are more questions than answers. Not much is known about the effects of biochar on the productivity of agricultural soils in Sweden. It is expected that biochar would increase the productivity on sandy soils due to its improved water holding capacity, but this is yet to be verified during controlled field experiments.

Even though biochar is very resistant to degradation, there are indications that biochar would stimulate microbes and thus the decomposition of soil organic matter. As a consequence, it is important to monitor the soil carbon dynamics in soils amended with biochar.

There is a need for additional studies quantifying leakage of nutrients from different soil types, especially studies comparing energy crops to food crops.

2.4 BIODIVERSITY

The importance of land-use changes for biodiversity and the importance of biodiversity for the natural capital and ecosystem services highlight a risk involved in the environmentally motivated bioenergy production. Any change in land use to produce bioenergy may speed up the loss of biodiversity, thereby counteracting its ultimate purpose to deliver sustainability.

This development is particularly troublesome in a global setting, where plantations of bioenergy crops often expand at the expense of natural forests or small scale farmland, thereby jeopardizing biodiversity and local livelyhoods (see below). In Sweden, the situation is somewhat different. Here biodiversity and human land use has to a large degree co-evolved, and accordingly, managed landscapes may have a high biodiversity and provide important ecosystem services. For example, some of the most species-rich habitats to be found in the country are semi-natural grasslands, which have been used extensively for biomass production over a period of several hundred years. Some forest types have a long history of land use, and certain harvesting methods in these forests may present better options for biodiversity than strict preservation. As described above, biodiversity is a matter of land use intensity, as opposed to a matter of natural *versus* managed land. Hence, the effects of biofuel production on biodiversity depend on the production system chosen and the intensity with which it is applied.

In this chapter, the biodiversity effects of land-use changes brought about by Swedish biofuel production have been reviewed. Due to the complex nature of the field coupled with the concise character of the text, the effects on biodiversity are only qualitatively assessed. Quantitative assessments may be found in the literature cited but are, however, lacking for most production systems and species groups. The effects on the field or stand level are mainly described; merely reflecting available literature (the need for landscape scale studies are commented on under section 2.4.2, Knowledge gaps, below). The review selectively focuses on the most important production systems for biofuels in current Swedish forestry and agriculture, including contemporary systems that are not common but are nonetheless considered to have large potentials to deliver bioenergy feedstock in harmony with ambitious biodiversity targets.

2.4.1 Effects on Biodiversity of Biofuel Production Systems

The higher outtake of biomass from forests may present a challenge to biodiversity. Effects may be caused by e.g., intensification, extraction of logging reidues and new types of plantations.

Intensification

The increased demand for biofuel production may work as an engine to intensify current production systems in Swedish land use. Standard measures to increase the production within forestry and agriculture include the use of fertilizers and pesticides, draining, use of fast-growing exotic species, utilizing residues such as branches or straw, and the harvesting of previous biodiversity set-asides. In most cases, such land-use changes involve major threats to biodiversity, since they change the basic ecological conditions of the ecosystems, i.e. replacing previous processes and structures by new ones, to which few ecosystem-specific species are adapted.

For example, the fertilization of forest changes the species composition of vascular plants, bryophytes, fungi, and lichens (Zetterberg *et al*. 2006, Strengbom & Nordin 2008, Forsum 2008, Dahlberg *et al*. 2010). The general effect on plant diversity is negative, and the effects on the understory vegetation may be long-lasting (e.g. Strengbom & Nordin 2008). Fertilization has been described as a threat to mycorrhizal fungi (Dahlberg *et al*. 2010). Although the effects are poorly investigated (Nohrstedt 2001), also larger species, such as birds and mammals, may be affected. Fertilization also impacts soil parameters (see section 2.3)

As another example, the introduction of exotic tree species is negative for biodiversity because they harbor only restricted proportions of the tree-depending species compared to indigenous trees (Miljøministeriet *et al*. 2008, Larsson *et al*. 2009, WWF 2009). Exotics may even be invasive and outcompete indigenous species, thereby threatening biodiversity on larger areas (Miljøministeriet *et al*. 2008, WWF 2009, Dauber *et al*. 2010).

Another way of intensifying production that may be either negative or positive for biodiversity is increasing harvesting frequency. A shortened rotation period in current industrial forestry is generally negative, because many forest-residing species are strongly depending on the continuity of the forest ecosystem and on structures and processes in old-growth forest, such as old trees and dead and decaying wood. On the other hand, many threatened species may be positively affected by the frequently repeated harvest of herbaceous plant tissue in semi-natural grasslands or of young wood tissue derived from shooting in semi-natural coppice. Harvesting of such ecosystems has a long tradition in Swedish land use and, accordingly, much of the flora and fauna have adapted to this pre-historic land use.

Extraction of logging residues 1: slash (i.e. branches and tops ="grot")

Slash contributes to forest biodiversity by providing substrate and habitat for many wood-residing organisms, including fungi, lichens, mosses and arthropods (reviewed in Naturvårdsverket 2006 and by Berglund 2012). Considering that almost all forest biomass is removed at clear-cutting, further substrate reduction through the removal of logging residues may break the last remaining substrate continuity in the production forest. For some rather generalist species on wood and litter, e.g. species of bryophytes, the extraction of logging residues has been shown to be negative because of reduced substrate abundance, reduced protection against drying-out clearcuts, and increased disturbance frequency (Gustafsson 2004, Åström 2006). The removal of slash can also affect the structural diversity favoring small mammals and breeding birds (Naturvårdsverket 2006).

However, recent research points out that the effect of spruce slash removal on threatened forest biodiversity is limited, because most of the more demanding species are lost already at logging. Slash from conifers generally harbor few demanding and red-listed species (Dahlberg & Stokland 2004, Junninen *et al*. 2006, Jonsell *et al*. 2007, Caruso *et al*. 2008). Slash removal does not seem to cause any significant changes in species diversity among soil fauna or fungi and is, therefore, not expected to have any significant effect on soil processes, such as mycorrhizae and decomposition (Naturvårdsverket 2006). The removal of spruce slash appears to have no adverse effect on the diversity of wood-residing fungi or lichens.

On the other hand, slash from oak, aspen and other broad-leafed deciduous trees is important for many wood-residing fungi (Nordén *et al*. 2004), and for a number of red-listed species (Cederberg *et al*. 2001, Jonsell *et al*. 2007, Hedin *et al*. 2008). The amounts of slash from deciduous trees is decreasing in Sweden (Olsson *et al*. 2011), and the extraction of slash from these species may have strong negative effects on the threatened biodiversity (Naturvårdsverket 2006, Berglund 2012).

Not is only the slash removal *per se* a problem for biodiversity. Piles of deciduous forest residues attract wood-residing insects, especially if the surrounding forest land is cleaned of dead wood, and the piles may then function as ecological traps for specialized and endangered species (Jonsell & Hedin 2009, Hedin *et al*. 2008). When extracting logging residues from clearcuts, wood left for biodiversity after cutting is frequently collected (Andersson 2000, Gustafsson 2004, Rudolphi & Gustafsson 2005). In addition, the extraction of slash increases the damage to the forest floor by heavy vehicles by 60% (Gustafsson 2004, Naturvårdsverket 2006), and remaining logs may be damaged leading to poor substrate quality for wood-residing insects (Hautala *et al*. 2004, Naturvårdsverket 2006, Berglund 2012).

Extraction of logging residue 2: stumps

Stumps are important as substrate for mosses, lichens and fungi (Naturvårdsverket 2006, Caruso *et al*. 2008) and plants that are sensitive to desiccation may decrease on stump-harvested clearcuts. Stump-harvesting has a long-lasting negative effect on blueberry production (Berglund 2012). Stumps make up a large proportion of the coarse dead wood left after forest cutting (Egnell *et al*. 2006), and may function as a refuge for a range of forest species depending on coarse dead wood, for example beetles and fungi (Naturvårdsverket 2006, Berglund 2012). An increase of stump harvest may, therefore, have negative effects on species that are presently common. The effects of the systematic harvest of stumps on biodiversity are, however, not well known.

Similar to the extraction of slash, piles of deciduous stumps may attract wood-residing organisms and function as ecological traps (see above). At stump harvesting, most of the remaining dead wood, such as logs left for conservation purposes, are either collected or destroyed (Andersson 2000, Gustafsson 2004, Rudolphi & Gustafsson 2005, Hjältén *et al*. 2010).

Extraction of logging residues 3: ash recycling

Use of logging residues affects biodiversity indirectly through the increased need for fertilization, e.g. through ash recycling. Application of ash to forest is also a way of managing the waste produced by wood combustion (Stupak *et al*. 2007b), and can, therefore, be considered an unevitable part of using logging residues for energy purposes. Different studies of the effects of ash recycling on ground vegetation, fungi and soil organisms show contradictory results, and the output may depend on site and dosage (reviewed by Aronsson & Ekelund 2004). Wood ash recycling may have negative effects on dwarf shrubs, mosses, liverworts and lichens (Kellner & Weibull 1998, Jacobsson & Gustafsson 2001, Dynesius 2012), but may also cause an increased abundance of grasses and herbs (Olsson & Kellner 2002, Moilanen *et al*. 2002), and a general increase of an abundance of soil organisms (Nieminen *et al*. 2012). Accordingly, just like other fertilization, ash recycling can lead to a changed species composition of forest flora and soil biota.

Willow coppice on farmland

The impact of willow coppice on biodiversity depends on alternative land use and the surrounding landscape. Willow coppice on arable fields has a higher abundance and diversity of vascular plants, earthworms, ectomycorrhizal fungi, insects, birds and mammals, compared to agricultural crops, spruce plantation or fallow soils (reviewed by Weih 2006 and by Helldin *et al*. 2010). Especially in an otherwise homogenous agricultural landscape, willow coppice can benefit many species (Skärbäck & Becht 2005); a proportion of 10–20% of coppice in open farmland has been estimated to be optimal for a number of bird species (Göransson 1994). On the other hand, species linked to the open agricultural landscape, such as certain birds, may decrease (Skärbäck & Becht 2005). When comparing willow coppice to woodland habitats, most available studies report a lower richness of species in the energy crops or no significant differences (reviewed in Dauber *et al*. 2010). The diversity of small mammals is lower in willow coppice than in small biotopes, such as shelterbelts, grassy ditches or canal banks (Reddersen *et al*. 2005).

The proximity to stands of natural forest or mature indigenous deciduous trees improves the conditions for both plant and animals to spread into willow plantations (Hoffmann & Weih 2005, Weih 2006, Baum *et al*. 2009). The biodiversity value can probably also be promoted by increasing the structural diversity of the plantations, for example by limiting their size, by allowing undulating edges, and by not harvesting the entire plantation on a single occasion (Hoffmann & Weih 2005, Weih 2006, Schulz *et al*. 2009, Dauber *et al*. 2010). Using a mix of willow varieties or species may also improve the conditions for biodiversity (Weih 2006, Schulz *et al*. 2009). Increasing the time cycles, both between harvests and between replacements of plants, will allow biodiversity to develop in the coppice, but may at the same time be disadvantageous to light demanding species (Weih 2006, Baum *et al*. 2009, Dauber *et al*. 2010).

If grown on farmland outside arable fields, the impact on biodiversity would in most cases be negative. Well-managed semi-natural grasslands, such as meadows and pastures, have a high biodiversity, which will obviously change drastically if planted with coppice. Compared to a ceased grassland management, e.g. spontanous afforestation or active spruce plantation, willow coppice may well prove to be better for biodiversity in general. However, the issue appears not to have been studied.

Populus plantations

Similar to the case of willow coppice, plantations of poplar and hybrid aspen will impact biodiversity depending on the alternative land use and surrounding landscape. When planted on previous arable fields, poplar and hybrid aspen plantations will probably gain in biodiversity, with a floristic diversity comparable to that in average mixed-wood forests, although most of the newly established flora will consist of common species (Weih *et al*. 2003, Soo *et al*. 2009, Strengbom 2009). The diversity of vascular plants, however, depends on ground treatment in connection with planting; ploughing and not least herbicide treatment will decrease the diversity (Soo *et al*. 2009, Strengbom 2009).

Compared to forest sites, the effect on biodiversity is not obvious. Hybrid aspen appears to be better for biodiversity than other types of intensive forest production, such as Norway spruce or logdepole pine (Gustafsson *et al*. 2009). Hybrid aspen probably harbours a similar insect fauna as common aspen, and whether hybrid aspen stands will develop a high insect diversity will likely depend more on what abundance of old trees and dead wood is allowed (Lindelöw 2009).

As with willow coppice, poplar and aspen plantations in the homogenous agricultural landscape introduce diversity on the landscape scale, which may benefit many species (Skärbäck & Becht 2005). Increasing the structural diversity of new plantations, for example by keeping down the size, by allowing undulating edges, and through asynchronous harvesting within and between stands, may further promote biodiversity (Hoffmann & Weih 2005, Weih 2006, Schulz *et al*. 2009, Dauber *et al*. 2010).

Caution is raised concerning the risk of hybrid aspen seed dispersal into new areas, or the spread of pollen/DNA into the common aspen population (Strengbom 2009).

Semi-natural coppice and pollarding

Current biofuel production based on coppicing and short-rotation forestry has so far been entirely restricted to willow, and to some extent poplar, on arable land. Its historic predecessor is semi-natural coppicing or pollarding. Stump or root sprouts of deciduous trees are harvested before they reach larger dimensions, often with a harvest cycle of 5-30 years, to maximize biomass production. Unlike willow coppicing, the historic semi-natural coppicing was based on various indigenous tree species, allowed stumps (“sockets”) to become very old, did not use fertilizers or pesticides, and only rarely took place on land that might be used for food production. A variant is pollarding, i.e. cutting shoots from higher stumps, normally above what can be reached by browsing animals. Another variant is coppice with standards, where single trees are allowed to grow tall, for timber production or conservation purposes. These types of forest harvesting were extremely widespread in the Eurasian nemoral and boreo-nemoral zones, as well as in the sub-alpine birch forest (Emanuelsson 2009, Helldin *et al*. 2010), but is today largely replaced by tall-tree production.

Although few systematic studies have been conducted on semi-natural coppice as a production form or habitat, it can be expected to have a rich biodiversity, to a large extent depending on the cutting and extraction of wood. It is considered to favor a wide range of rare insect and plant species (Kirby 1993, Key 1995, Rydberg & Falck 1996, Broome *et al*. 2011), as well as birds and small mammals. The ground flora in semi-natural coppice resembles that of tall deciduous forests, but includes light demanding species which are favored by regular cutting (Staun & Klitgaard 2000). The frequent harvest of shoots creates microhabitats and a general habitat structure that may be highly beneficial to threatened biodiversity, e.g. species connected to sun-exposed substrates or a stable supply of dead and decaying wood (Helldin *et al*. 2010). Pollarding favours a diversity of often rare lichen, and wood beetles of high conservation priority (Hultengren *et al*. 2006, Dubois *et al*. 2009), and pollarded trees are also important to wood fungi, epiphytic mosses, and species depending on tree cavities, such as hole-nesting birds (Dagernäs 1996, Moe & Botnen 2000). Coppice with standards could potentially favour biodiversity connected to sun-exposed old-growth trees, for example oak and pine, which are today threatened by shading in formerly semi-open stands (Helldin *et al*. 2010).

Several authors have proposed semi-natural coppicing to be resumed for nature conservation purposes (Fuller & Warren 1993, Kirby 1993, Gustavsson & Ingelög 1994, Rydberg & Falck 1996, Staun & Klitgaard 2000, Otte *et al*. 2008), and this is conducted mainly on the British Isles (Rydberg & Falck 1996). Small scale pollarding is supported by the Swedish program for agri-environmental subsidies and hence conducted in many parts of Sweden. Only in recent years, this type of logging has been identified as interesting to bioenergy production on a significant scale in harmony with biodiversity targets (Helldin 2008, Schaber-Schoor 2009, but see Rydberg & Falck 1996).

The components of semi-natural coppice and pollarding can be introduced to current energy forest production to improve the conditions for biodiversity. Such components include, for example, the diversity of tree species, using indigenous species, letting stumps develop into "eternity sockets", and establishing coppice in what is considered to be forest areas.

Ley

As with most agricultural crops, the production of grass from ley has no obviously positive biodiversity effects. Leys for bioenergy production are often intensively managed, and the fauna and flora are accordingly species poor (Isselstein *et al*. 2005, Plantureux *et al*. 2005, Prochnow *et al*. 2009a). The main positive effect is that leys contribute positively to habitat variation in agricultural landscapes dominated by annual crops (SLU 2007). Conversely, in landscapes already dominated by ley and similar non-intensive cultivation (often in forest regions), a further increase in the proportion of ley has mainly negative effects (Wretenberg *et al*. 2006, SLU 2010). Ley *per se* is somewhat more attractive to birds than high-intensive annual crops (Berg 1992, Berg & Kvarnbäck 2005, Berg & Gustafson 2007). A similar advantage has been suggested to insects, especially in clover fields, but this advantage is doubtful considering the early harvest of leys (SLU 2007).

However, when using ley for energy purposes rather than for fodder, the number of harvests each year can decrease and the date of the first harvest be postponed (Møller *et al*. 2007 and unpublished), both of which may benefit breeding birds, mammals, and nectar-feeding insects. With a more diverse species mix sown or some important nectar species added (e.g. *Lotus corniculatus* or Cicoria), the diversity of beneficial arthropods, such as pollinators and natural enemies to pest species, can increase (Gardiner *et al*. 2010, Borgegård pers. comm.)

Phalaris and other perennial feedstock

Few studies have been conducted on biodiversity in *Phalaris* or other perennial crops grown specifically as feedstock. The few studies available indicate that fields of *Phalaris* feature a higher abundance of ground flora, birds and small mammals, and feature a higher arthropod diversity, compared to average annual crops (Semere & Slater 2007a, 2007b). In general, perennial biomass crops are perceived as being beneficial to biodiversity compared to cultivated areas of arable food crops because biomass crops have longer rotation periods, less chemical input, better soil protection, a greater richness of spatial structures, allows harvesting to be carried out after the breeding season for birds, and are also exposed to fewer disturbances during the growing period (Dauber *et al*. 2010).

Wetland grass

Open semi-natural wetlands in the agricultural landscape are generally important to biodiversity, but during the last century, there has been a major loss of traditionally managed wetlands, due to either intensification (drainage and cultivation) or abandonment (ceased mowing or grazing). Using wetland grass harvested for nature conservation reasons as feedstock for bioenergy provides an opportunity for the resumed management of wetlands to the benefit of biodiversity. This potential has been highlighted in a series of recent papers (Peeters 2009, Prochnow *et al*. 2009a, 2009b, Roesch *et al*. 2009, Foster *et al*. 2009). The agri-environmental subsidies make wetland management economically interesting despite the rather costly harvest and the lower biogas yield compared to haylage (Florell, Swedish Biogas International AB, pers. comm).

The large positive effects of the resumed management of former wet meadows are well documented, for example for birds and carabid beetles (e.g. Alexandersson *et al*.1986, Ljungberg 2001). At present, no disadvantages to harvesting grass for biogas are known, compared to the commonly performed mowing or grazing for conservation purposes. The problem of premature harvesting to wetland birds need to be considered in relation to the quality demands of the biogas production.

Grass from other semi-natural grasslands

Drier semi-natural grasslands need to be regularly harvested to retain their biodiversity, and the harvest can potentially be used as feedstock, either for biogas or for combustion. Present management of such land is dominated by livestock grazing, albeit cutting of hay was historically the dominating form of land use; hence, resumed cutting and grass removal could be the best management method for biodiversity. Many previous semi-natural types of grassland are also abandoned or planted with forest. As with wetlands, agri-environmental subsidies could make it profitable to harvest these grasslands for bioenergy. The practice is not common in Sweden, but is practiced on the continent, e.g. Germany, where extensively managed land is harvested primarily for biodiversity or for its landscape character (Prochnow *et al*. 2009b). It is proposed as an alternative to corn as feedstock production on American prairie soils. The potential for using semi-natural grass in

Sweden is regionally high. The basic ecology of these ecosystems is well known, but further knowledge on harvesting practices to optimize biodiversity and economy is needed.

2.4.2 Knowledge gaps

The effects of the extraction of logging residues on biodiversity have been addressed within the Swedish Sustainable Supply and Refining of Biofuels Research Program running 2007-2010 (de Jong *et al*. 2012). However, certain aspects need further elucidation, for example the preference of stump size for many wood-residing insects and lichens, the impact of ecological traps on red-listed species, the effects of the extraction on vertebrate taxa (amphibians, reptiles, small mammals and birds), and the potential impact on organisms that are presently common.

Biodiversity in willow coppice has been studied in a number of previous projects, but questions remain concerning willow coppice grown outside arable fields. If plantations of poplar and hybrid aspen should become more common, the biodiversity effects of these crops need to be studied further. For all perennial bioenergy crops, it seems reasonable to develop means of adapting the detailed management to optimize biodiversity, for example by finding the optimal harvesting time and frequency, treatment of the feedstock, cutting hight, species mix, pest treatment, size, shape and turnover time of fields/stands, and the siting of plantations in a landscape context.

It appears particularly important to initiate studies of the potential of "win-win production systems", i.e. feedstock derived from biodiversity-friendly harvesting in semi-natural habitats (grassland or coppice). Such research would give an indication of whether these systems are worthy of investments. The research needs to address the various aspects of optimizing the management for economic and ecological sustainability, e.g. the timing of the harvest, the treatment of feedstock, monetary values of ecosystem services, and conditions for combining it with other land use, such as outdoor recreation or livestock grazing. A certain amount of technical development is needed to find large-scale harvesting methods that reasonably mimic the harvesting that has been conducted historically in these habitats.

Analyses and assessments of landscape scale effects are needed (Dauber *et al*. 2010). Currently, we have a limited understanding of the proportion of land covered by energy crops that would significantly affect species richness or population viability. If landscapes to study this are lacking, a modelling approach can be adapted. Such modelling would include geographically explicit analyses of habitats that may be changed or replaced by energy crops, and the total area and location of the remaining habitat patches in relation to species area requirements and landscape fragmentation. It would also include analyses of energy crop fields/stands in relation to other habitats, and how these interact, for example by providing ecotones, additional habitats, landscape connectivity or ecosystem services (such as pollination or natural enemies).

2.5 SOCIOECONOMIC ASPECTS ASSOCIATED WITH LAND-USE CHANGES

There is a strong political and societal pull for increased use and production of biofuels both to make a transition away from fossil based fuels, but also to create a greater energy self-sufficiency in many countries (Chum *et al*. 2011; UNECE and FAO 2011; BP 2012; EC 2012; Johnson *et al*. 2012; White House 2012). Biomass for the production of biofuels can be derived rom all types of land; farmland, forests as well as from the sea (aquatic biomass). Most of the liquid biofuels produced in Sweden today has their origin in biomass produced on farmland. Biofuels based on forest products exist but in relation to other volumes, they are minor (Energimyndigheten 2012). Cellulosic based processes are, however, considered to open up for increased volumes of biofuels and the possibility of using feedstocks not commonly found today.

An increased use of biomass from the different ecosystems will have a range of effects. This presentation will depart from the conceptual framework of ecosystem services. This framework was developed in cooperation with ecologists and economists in order to be able to describe the uses society have of the services obtained from the ecosystems (Millennium Ecosystem Assessment, 2005)[2]. In this approach, services received from the ecosystems are grouped into four categories: i) provisioning services, ii) regulating services, iii) cultural services, and iv) supporting services. These four categories of services are linked to aspects of human well-being (Figure 6).

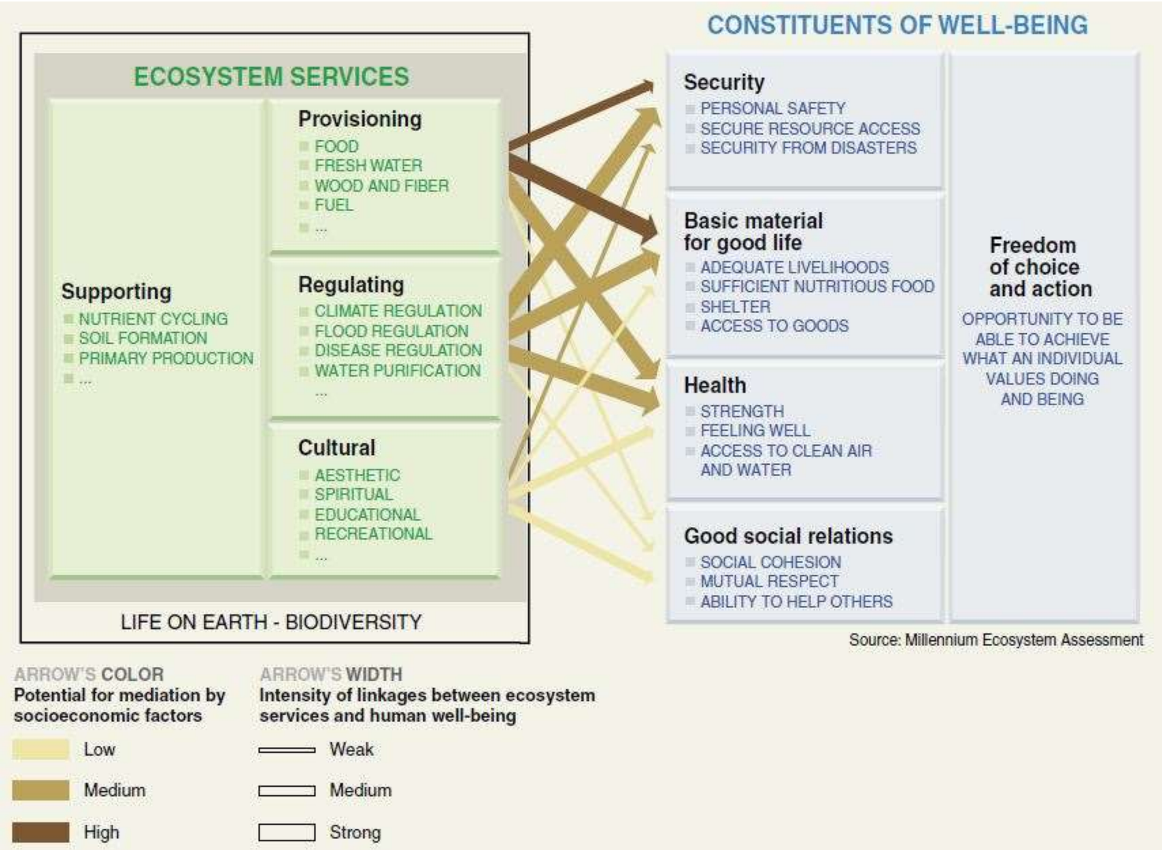

Figure 6: Millennium ecosystem services and links to human well-being (Millennium Ecosystem Assessment 2005).

Typically, the provisioning services are in focus which in our case would include the actual biomass produced on a specific piece of land. Regulating services, such as erosion control and buffer capacity, are also included, as well as supporting services such as the pollination as part of biodiversity in the landscape. In addition, there are a range of cultural services, such as possibilities for recreation and the value of the landscape. A land-use change or change in intensity of the produc-

[2] The need to marry economy and ecology was discussed already in the late 1980s (Costanza and Daly, 1987; Daly, 1990).

tion on a specific piece of land will affect both positively and negatively these four categories of services obtained from the ecosystems.

Ever since the framework of ecosystem services was presented, one of the challenges has been to quantify the ecosystem services in order to be able to make a comprehensive and inclusive analysis. The problem is that many of the services are difficult to value in monetary terms as the experience of the landscape would vary from individual to individual.

In 2010, a report was presented by UNEP (TEEB 2010) in which an approach for evaluating ecosystem services was introduced in order to make strategic decisions based on these services. Another report that is aimed at supporting business operations to consider ecosystem services in decision-making is Hanson *et al*. (2012). In Sweden, quantifying environmental changes in monetary terms is commonly found. An introduction to the field is found in Kinell *et al*. (2009). One of their conclusions is that Swedish case studies are too few in number in order to make conclusions on typical or default monetary values linked to changes in ecosystem services. To further complicate matters, the results are highly contextualized and linked to the specific setting in which the results are originating which makes generalization difficult.

In the following chapters, a presentation is made on some possible effects that an intensification of biomass production and/or land-use change may have on ecosystem services. A similar approach has been used by Gasparatos *et al*. (2011) when discussing the impacts of the expansion of biofuel production in Southern countries.

2.5.1 Forests – socio-economic aspects of changes in ecosystems services

An intensified forestry will bring about a number of changes in forest ecosystems. The intensification is not a land-use change, but rather a change in land use management. A recent review initiated by the Swedish Energy Agency was looking at the state of knowledge of the environmental consequences of Swedish forestry (de Jong and Lönnberg 2010; Akselsson. 2012). The study departed from the increased intensification of forestry linked to realizing the 16 Swedish environmental objectives. Its conclusion was that a certain level of increase in forestry could be acceptable considering the environmental objectives and certain conditions under which compensatory measures, such as ash recycling and forestry controls and regulations, were followed. The word “acceptable” is important, as it is a subjective judgement and also indicates that there will be negative impacts on some ecosystem services, while others would experience positive impacts. A number of environmental organizations in Sweden (Swedish Society for Nature Conservation and WWF Sweden among others) would also argue that the current Swedish Forestry Act is not operational in securing necessary respect for biodiversity and other environmental values in the forests (Sahlin 2010; Sahlin 2011; WWF 2011).

The Swedish forestry is regulated and based on two legs: i) production goals and ii) environmental considerations. These legs link to various categories of the ecosystem and will benefit different stakeholders of Swedish forests. For example, forest production will benefit the owner, the water regulating service will benefit communities in the watershed area and biodiversity values would benefit future generations. The Swedish forestry debate has in recent years been expanded to include a much broader segment of stakeholders. As a matter of interest, the management of privately owned forests has been subjected to the opinions of outside stakeholders who may not even recognize their roles as discussion partners[3]. With opportunities provided by for example social media to spread messages and create public opinion, the identification of stakeholders is much more complex today than some years ago. The activities of this expanded stakeholder group can be illustrated by a series of articles in the Swedish newspaper DN on the state of Swedish forestry and subsequent replies from the forestry sector and other stakeholders[4]. The reporter stated that the forests have become an industrial landscape with production as its overarching aim and that forests were no longer accessible to the population. The forestry sector contradicted this viewpoint. Thus, the forestry sector is no longer closed to fresh perspectives brought in by new stakeholders (Beland-Lindahl 2009; Beland-Lindahl and Westholm 2010). For example, the actors looking at forestry as a carbon sink will bring this "new" variable into the discussion of forest management and biomass use. One cannot both cut down a tree and keep it; hence, there is a certain degree of conflict in the production goal *vis a vis* environmental objectives.

Provisioning services - forest

Examples of provisioning services are food, fresh water, fuels, wood and fibers. Increased outputs from forestry will provide higher economic returns in the short-run. The intensification of the forestry will lead to higher growth rates and links to the overarching aim of production from the forestry. An increase in standing biomass has been monitored in Swedish forests since at least 1950 (SLU 2011). In addition, there are a range of measures that can be taken to further increase growth rates (Ståhl 2009). Climate change is anticipated to affect growth rates as well as ecosystems and biodiversity (Olsson 2011).

There are many people involved in forestry who get their main income from working in this sector. As a consequence of mechanization and changes in forestry management (see Figure 7), the number of hours of productive work in the forests has decreased during the last decades. The volumes of fellings have increased over the same time period (Skogsstyrelsen 2011).

---

[3] A stakeholder is an individual or group that has an interest in any decision or activity of an organization ISO (2010). Guidance on Social Responsibility. Geneva, International Organization for Standardization (ISO).

[4] The full list of articles: (Jonsson and Thorstensson 2012; Ledare 2012; Lundmark *et al* 2012; Salin 2012; Stridsman 2012; Zaremba 2012a; Zaremba 2012b; Zaremba 2012c; Zaremba 2012g; Zaremba 2012f; Zaremba 2012e; Zaremba 2012d; Åström and Terstad 2012).

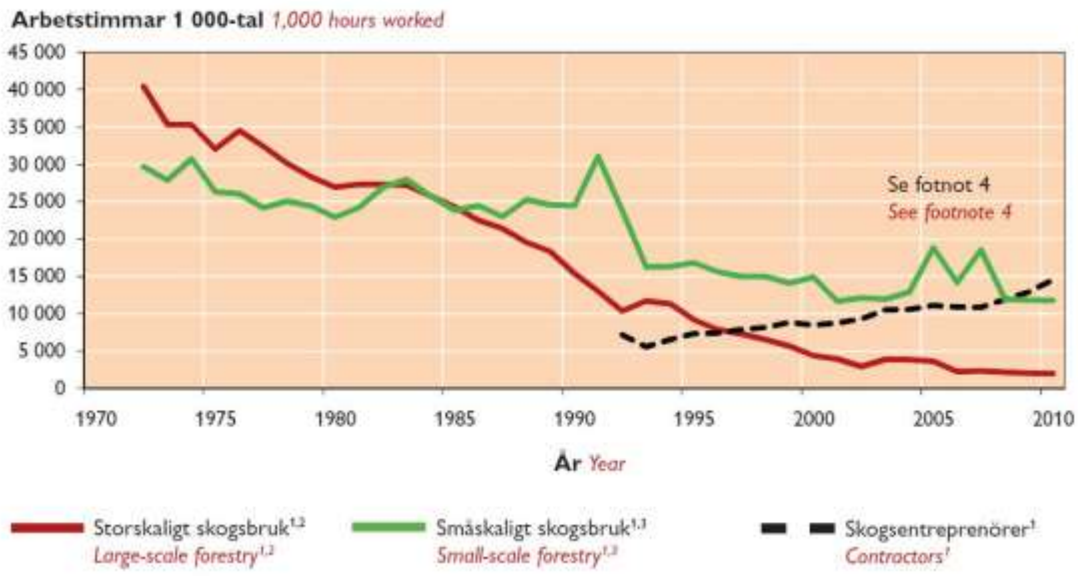

Figure 7. Working hours in forestry (Skogsstyrelsen 2011).

Other provisioning services from the forests are fruits, berries, and mushrooms. To many people this is an important source for income. Most people involved in commercial berry picking will be seasonal workers coming from countries other than Sweden (Jonsson and Uddstål 2002). In addition, hunting with more than 250,000 hunting licenses (not necessarily linked to forests) each year is a large activity. In 2008 the value of hunted animals in Sweden reached about SEK 3 billion (Mattsson *et al*. 2008). This can be compared to the gross value of timber felled in Sweden during 2009, SEK 23.3 billion, and to the total value of the whole forestry industry, SEK 216 billion (Skogsstyrelsen 2011). These examples show that much of the value generated from the forests do not derive from the actual biomass in the trees, but from other plants and animals living in the forests. The possibility to access these provisioning services is described in the *right to public access* ("allemansrätten"), a customary right that is highly valued by the Swedish population (Sandell and Svenning 2011).

Regulating services - forest

The ecosystems in the forests provide a number of regulating services, for example climate regulation, flood regulation, disease regulation and water purification. At present, there is an on-going debate on how to calculate and quantify the carbon sequestration capacity, as well as the storage of carbon in standing biomass over time (see e.g., Repo *et al*. 2010; Dehue *et al*. 2011; Zanchi *et al*. 2011; Manninen *et al*. 2012). This issue is linked to land-use change and indirectly to land-use change effects and is further discussed in Chapter 3.2.

Other regulating services include erosion control and buffering capacity. An intensification and increase of output of biomass from forests will affect both aspects. Concerns have been raised about the increased levels of run-off with nitrogen and methyl mercury as a consequence of intensified forestry. Such intensified forestry could also mean controlling water tables.

Supporting services - forest

Examples of supporting services include nutrient cycling and soil formation. In Sweden, clear cutting became the dominant silviculture method for harvesting the forest biomass in the 1960-1970's. During these years, there was only slight formal attention paid to preserve environmental values, a situation which has changed since. Still, the clear cut of an area will have immense effects on the ecosystems, flora and fauna. Measures are made to decrease the negative environmental impacts resulting from clear cuts. One such measure is introduction of retention forestry, leaving some trees or groups of trees in the clear cut area (Gustafsson *et al*. 2012).

A number of environmental organisations would like to see a higher share of so-called "continuous cover forestry" (Johansson *et al*. 2009; Naturskyddsföreningen 2011). Continuous cover forestry will typically reduce negative impacts on the ecosystems and support high biodiversity in the forests, a management regime that should be considered an alternative method to clear cutting (Dahlberg 2011).

Biodiversity can also be affected by the introduction of non-native species to the ecosystems. In Swedish forestry, the introduction of the Lodgepole Pine (*Pinus contorta*) has raised concerns of the Swedish pine tree species will be affected.

Cultural services - forest

The forest provides a number of cultural ecosystem services connected to aesthetic, spiritual, educational, and recreational values. Recreational values and being able to freely move about to experience nature are considered to be of great value to the population and it is a customary right for Swedish people to enjoy access to forests and other land. The research project "Friluftsliv i Förändring" (*Outdoor Recreation Undergoing Change*) focuses on changes in how people are making use of nature for recreation and other activities (Fredman *et al*. 2006). The research project illustrates the gap of knowledge on how to value recreation and the role of nature for the general well-being of people. In a recent thesis by Sandberg (2012), the issue of children's interaction with nature is analyzed. He finds that parents in general would like to see their children spend more time outdoors, for example in forest areas close to where they live, and that the experience the children receive from their outdoor activities will be central in their upbringing. These examples highlight that there is a redefinition of the value that forests, farmlands and other places have to human beings. In a pilot-study by Sonntag-Öström *et al*. (2011), it is shown that people with stress-related exhaustion are feeling better after being exposed to boreal forests.

It is difficult to assess whether recreational aspects would be affected by an intensification of biomass production in the forests. Forests are considered "beautiful" and easy to access. Other aspects relate to accessing provisional services, such as berries, mushrooms and hunting. At the same time, many provisional services go hand in hand with recreation – for example, people like to spend time in the forests since they are relaxing, frequently in combination with picking mushrooms.

The way forestry is carried out affects the scenery and image of the landscape and how it is perceived by the public. A clear cut area where the stumps have been retracted will display a landscape different from other clear cut areas. The experience of the cultural landscape is an aspect that many people cherish and do not wish to see altered. An example is the controversy concerning the location of wind power plants in the Swedish landscape (Henningsson *et al*. 2012), where many argue that these plants disturb the visual enjoyment of the landscape.

2.5.2 Socioeconomic aspects of bioenergy from farmland in Sweden

Biofuels, ecosystem services and human wellbeing

In a review of the drivers and impacts of biofuel production on ecosystem services and human wellbeing, Gasparatos *et al* (2011) present a framework based on the Millenium Ecosystem Assessment (2005). Ecosystem services are classified into provisioning, regulating and cultural services. The provisioning services of agriculture are food, but also fuels, and there is a trade-off between them. Another provisioning service is related to water, both quantity and quality. Important regulating services include climate regulation, air quality regulation and erosion control. Cultural ecosystem services provided by agricultural land include the open landscape, which is cherished in a country mainly covered by forests, as well as animals, especially horses. Biomass production on agricultural land can contribute to keeping the landscape open when crops are harvested every year. On the other hand, woody energy crops such as willow are sometimes seen as a threat to the open landscape. Cultural ecosystem services can have a large economic value, especially for tourism.

In the framework of the Millenium Ecosystem Assessment, ecosystem services are seen as affecting human well-being in a number of ways. The aspects of human well-being that are related to bioenergy from agriculture include rural development, energy security, food security, health, land tenure and gender issues (Gasparatos *et al*, 2011).

Socioeconomic aspects of agricultural land-use change

Swedish agricultural land use during the 1990-2010 period has changed; there are 30% fewer farms that occupy on the average 34% larger size (a change from 30 to 40 ha), which produce approximately the same economic total value (Edenbrandt, 2012). Farms larger than 100 ha now cover half the agricultural land. Larger farms experience higher growth rates. The main change in land cover is a trend towards large shares of grasslands. Farms have become more specialized in terms of agricultural products, especially small and medium-sized farms. However, farmers have become more diversified in their activities, with income from activities other than farming, such as tourism, transport, energy production and small scale food processing. Farmers that give up farming are either young or are approaching retirement. Middle-aged farmers are least likely to give up farming, possibly because of a lack of other opportunities. More farms have closed in the sparsely inhabited forest regions where farms are smaller. The regional specialization has become more pronounced, with Southern fertile plains becoming more specialized in crops and forest regions more specialized in cattle breeding. The development towards fewer, larger and more specialized farms is attributable to a number of factors: technological development, price development, demography, policies and a structural change among customers (Edenbrandt, 2012).

Experiences of farmland bioenergy in Sweden

There are a number of potential sources of biomass for biofuels from Swedish agricultural land, with differing socio-economic implications. The possibility of cultivating biomass for biofuels might provide new options for income in the forestry sector. At present, much of the biofuels seen on the Swedish market derive from crops such as wheat, rapeseed or barley (Energimyndigheten 2012), established food and feed crops that do not require any changes in land use or farm management.

The Swedish experience of willow plantations (Salix) short rotation forestry presents an interesting case for the socio-economic aspects of bioenergy from farmland. Willow plantation was promoted among farmers in the early 1990's and resulted in 16,000 ha of farms without livestock, mainly in central Sweden around Lake Mälaren (Dimitriou *et al* 2011, Rosenquist *et al* 2000). Subsidies were offered for the conversion to Salix. Despite projections for a drastic increase, the area planted slightly decreased and many farmers were disappointed with their Salix investments (Helby *et al* 2006). A main reason was that yields were not as high as expected because Salix was planted on low-quality land that was not well managed (Dimitriou *et al* 2011). Prices for Salix fuels have been lower than expected, but were not the main cause for this failure (Helby *et al* 2006).

Farmland biogas development has been constrained by poor profitability. To overcome this problem, a 30% investment subsidy was introduced in 2009, which led to a noticeable increase in farm based biogas. However, in the spring of 2012, this still added up to only 35-40 plants. To be profitable, manure based biogas plants would have to be larger than the medium Swedish dairy farm. There are technical solutions for relatively small, simple plants for single farms, but there are economic and production benefits related to larger plants which demand more resources in terms of capital, substrate and labor than managed by the typical Swedish farmer. Thus, there are potential benefits accruing to farmers who engage in collaborative ventures.

Straw as a farmland biomass resource has been investigated and discussed for 20 years but has met with only minor response in Sweden. In contrast, in neighboring Denmark, straw is an established biomass source used in small scale installations, as well as in district heating and CHP plants. Straw is a problematic fuel, but the Danish experiences show that the technical difficulties are possible to overcome. In a comparative study of the use of straw for heating in Denmark and Southern Sweden, Voytenko and Peck (2012) conclude that the success of straw bioenergy in Denmark was caused by "a combination of strong political will, adoption of effective policy instruments supporting straw use for energy, collective action [...] and extensive research in the field. All successful examples of transition towards bioenergy were found to deliver co-benefits, and collaboration between actors in the biomass chain in the form of contracts, agreements or through shared co-benefits are crucial to progress." (Voytenko & Peck 2012, p. 45).

Farm businesses and bioenergy

When considering the development of agriculturally based bioenergy from a farmer entrepreneurship perspective, a number of issues arise: Farmers have to recognize opportunities if they are to use agricultural resources (fields, machinery, labour, etc.) for bioenergy production in line with their attitudes and values. Shane & Venkataraman (2000) call this "opportunity recognition". Moreover, the opportunity must be perceived to offer a greater value than the value gained from the current use of resources (alternative costs), plus a risk premium. If this were to occur, the farmer would need to have both the knowledge and the ability to understand and recognize the value of such a new opportunity (Cohen & Levinthal 1990).

In addition to business aspects such as expected economic costs and returns, profit maximization, and strategic implications; the business development decisions of farmers are also influenced by social aspects, such as family duty, lifestyle, and community image (Hansson & Ferguson 2010). In a comparative study of 18 cases of decentralized bioenergy systems from the developed as well as the developing world, Mangoyana & Smith (2011) point out the key role of economic factors for driving small scale energy systems, as well as the importance of the wider community adoption of bioenergy use. The social context in which farmers exist implies that any new business development opportunity will depend on the interplay between the new technical opportunities and the values, attitudes, norms and rules of the social environment of farmers (Jack and Anderson 2002). In the case of choosing bioenergy crops, attitudes towards landscape values are important. Farmers as well as others cherish certain landscapes due to their perceived significance (Selman 2010), perceptions that may matter in their decision-making on which new crops to sow.

Agricultural policy and bioenergy

Agricultural policy affects bioenergy from farmland, especially the EU Common Agricultural Policy (CAP) and the EU Rural Development Policy (RDP). The RDP, with its focus on rural development and the environmental effects of production, is the so-called CAP "second pillar", while the "first pillar" concerns direct support payments to farms. The current RDP points to the importance of climate change mitigation, yet in the plans for the next CAP period (2014-2020), this is further emphasized (EC, 2011). Biomass and renewable energy production are seen as keys to unlocking the potential of the agricultural sector to mitigate climate change. However, it is unclear whether this potential will be realized (Waldenström *et al* 2011).

Within given limits, member states may choose to emphasize certain objectives and specific measures in their national RDP programs. During the present program period, the Swedish budget for axis 1 (*competitiveness of the agricultural and forestry sector)* is 14 percent of the total public RDP budget, the budget for axis 2 (*the environment and the countryside through land management)* is 69 percent, the axis 3 (*diversification of rural economy and improving the quality of life in rural areas)* budget is eight percent and the axis 4 (*local governance and territorial development)* budget the minimal seven percent. Thus, agro-environmental measures dominate the Swedish implementation of the program.

2.5.3 Knowledge gaps

A main challenge is to be able to account for and make initiated decisions about mutually accepted arguments. As shown in Beland-Lindahl (2009), the arguments are not shared among all stakeholders. Some arguments are agreed on by some stakeholders who may not agree on other arguments. As the forestry and bioenergy production on farmland have become part of the climate, energy and biodiversity debates, new stakeholders have ventured into these debates, bringing arguments and claims of ecosystems services that were previously not major issues in these debates. The discussion on the socio-economic aspects of forests and farmlands has not become less complex over time, but rather the opposite.

In Sweden, nature is accessible and free for all to experience within certain limits which makes a large portion of the population into stakeholders of the Swedish landscape. Among ecosystem services, recreational values loom large. The increase of biomass output from forestry and farming will affect the experience of the landscape, as well as the opportunity to access these lands. There is a need for further research on how various stakeholders and their interactions may contribute to and be affected by land-use change caused by increased biofuel production.

3 LAND-USE CHANGE EFFECTS: GLOBAL PERSPECTIVE

3.1 CLIMATE IMPACTS OF LAND-USE CHANGE

There are a number of ways in which changes in land use can influence the climate. For example, both warming and cooling effects occur at deforestation (Figure 8). Emissions of greenhouse gases from soil and the combustion of biomass lead to global warming effects. Also, deforestation leads to reduced evapotranspiration. “Evapotranspiration” is the collective term for water movements in a plant, and since it requires energy, the process has a cooling influence. This means that deforestation will lead to decreased evapotranspiration and a warming effect compared to keeping the forest intact.

A living forest emits organic compounds (aerosols) which can have a direct (by scattering sunlight) and indirect (influencing clouds) cooling effect on climate. Consequently, land-use changes such as deforestation implies a warming effect compared to keeping the forest intact. Aerosols are also formed when biomass is burned, for example when clearing the land for cultivation. These aerosols are thought to have a cooling effect on the climate. However, it should be kept in mind that the effect of aerosols on clouds, precipitation and the climate remains one of the largest uncertainties in our climate system (IPCC, 2007).

Forests are usually dark, meaning that deforestation will increase the albedo (whiteness), which increases the reflection of sunlight and, therefore, has a cooling effect. This effect is especially evident in areas where there is snow during the winter. Imagine a coniferous forest that is green all year around, compared to an agricultural field, which is white during the winter.

In the following chapters, the factors relating to land-use change and climate are further explored.

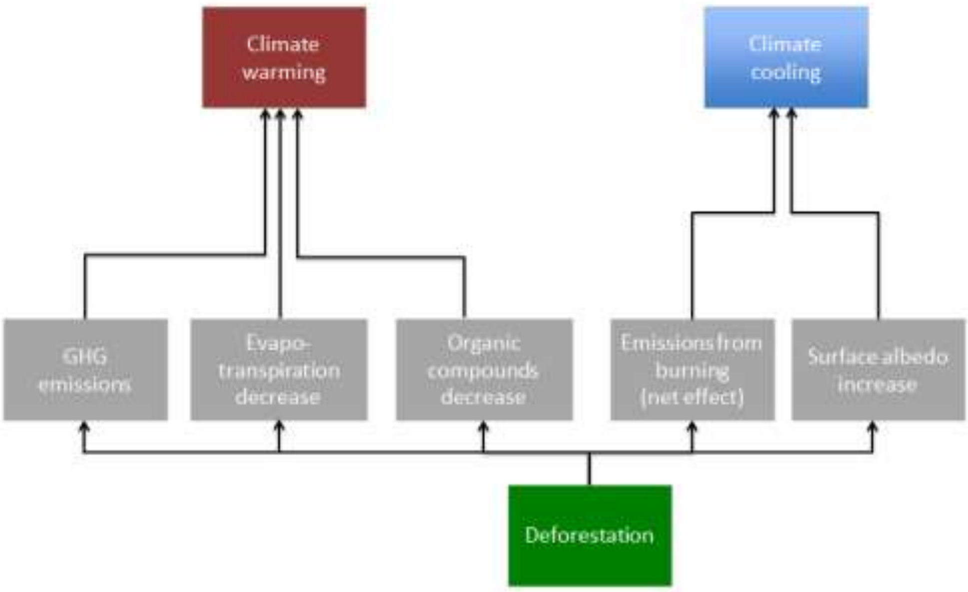

Figure 8. Simplified example of factors that influence climate due to land-use change, in this case deforestation. Inspired by Spracklen et al. (2008).

3.1.1 Different Methods of Quantifying Climate Impact

Before further exploring the factors behind climate and LUC, it is important to be aware of how we quantify climate impact.

About half of the incoming solar energy reaches the Earth's surface where it is absorbed by land and ocean. The rest of the solar energy is reflected back into space or is absorbed by our atmosphere. The energy which reaches the Earth's surface is radiated back into space as long-wave energy. The balance between absorbed and radiated energy determines the average temperature on Earth – the so-called radiation balance. Certain factors can influence the radiation balance. According to the IPCC definition, radiative forcing (RF) is a measure of the influence of a factor in altering the balance of incoming and outgoing energy in the atmospheric system of the Earth, and is expressed as Watts per square meter (W/m^2).

An increase in radiative forcing tends to warm the surface of the planet, while decreased forcing cools it. For example, increased concentrations of CO_2 lead to positive radiative forcing and a warmer climate. The increase or decrease in radiative forcing is measured relative to a reference level. The year 1750 is often used by IPCC as a reference, representing a pre-industrial level of human development (IPCC, 2007; Zetterberg, 2011).

The global warming potential (GWP) is a commonly used climate impact metric. It is a method to weigh together different greenhouse gases, using key factors (Table 2). The GWP factors reflect the ability of each greenhouse gas to trap the heat in the atmosphere relative to carbon dioxide. The GWP factor of a greenhouse gas is defined as “the integrated radiative forcing over a specified period (e.g., 100 years) from a unit mass pulse emission” (IPCC, 2007). It takes into account the lifetime and radiative efficiency of a substance in the atmosphere.

Table 2. GWP Factors (IPCC, 2007)

	20 years	100 years	500 years
Carbon dioxide (CO_2)	1	1	1
Methane (CH_4)	72	25	7.6
Nitrous oxide (N_2O)	289	298	153

This means that emission of 1 kg of methane is equivalent to emitting 25 kg of carbon dioxide, when applying a 100 year perspective. Since the reference gas used is carbon dioxide, the GWP is expressed as kg (gram or ton) CO_2 equivalents.

In most LCA of biofuels, the GWP factor for CO_2 emitted during combustion of the fuel is put at zero on the assumption that the same amount of CO_2 emitted was sequestered during the growth of the plant feedstock. For annual crops, this is a reasonable assumption. However, for feedstock with longer rotations, such as forestry or short rotation coppice, there is a time lag between emission and sequestration. For example, if the accounting were to start with the felling of a tree and the tree were combusted, there will be a release of CO_2 into the atmosphere. If a new tree were planted on the same location, it would take a full rotation (up to a 100 years) before the same amount of CO_2 would be withdrawn from the atmosphere. During that time, the CO_2 would have a climate effect. Similar land-use change will contribute both to the release and sequestration of carbon, which varies over time.

Several approaches have been proposed in order to account for the time dependent effects of bioenergy systems in LCA (Bird *et al*. 2009; IEA, 2010). Some alternative metrics include the GWP Bio Index (Cherubini *et al*. 2011), and the Global Temperature Potential (GTP) (Shine *et al*. 2005). The climate impact can also simply be expressed as the radiative forcing over time. In this way, the impacts of albedo, evapotranspiration and aerosols can be included.

However, it should be realized that RF is a metric quite different from the GWP, the latter being a policy metric that allows a comparison to be made between the marginal impact of the addition of the emission of a gas relative to CO_2. However, since GWPs use the radiative forcing of a gas, cloud or aerosol (IPCC, 2007), the RF concept underlies GWPs. Climate metrics, their science and policy applications have been discussed in great detail by Fuglestvedt *et al*. (2003). Fuglestvedt *et al*. conclude that the radiative forcing concept is a robust and useful metric of the potential climatic impact; however, even though GWP has its shortcomings, it has nonetheless some advantages in terms of political feasibility.

3.1.2 Greenhouse gas emissions from Land-Use change

Biomass contains significant carbon stocks in both above-ground and below-ground parts. Apart from the type of vegetation or ecosystem, in existence on the land, biomass carbon stocks are affected by the climate zone and geographical regions (see Table 3).

Table 3. Carbon stocks (t C/ha) in above and below ground biomass in different land types exemplified by some climate zones and geographical regions (from Carre et al (2009) based on IPCC default factors).

Land use	Region	Climate zone			
		Boreal	Cool temperature, wet	Tropical moist	Tropical wet
Grassland	All	4.3	6.8	8.1	8.1
Shrubland	Europe		7.4	n.a.	n.a.
	America	7.4	7.4	53	53
	Africa	7.4	7.4	46	46
Forest less than 30% canopy cover	Europe	n.a.	14	n.a.	n.a.
	Asia continent	12	n.a.	21	36
	S America	12	21	26	39
Forest above 30% canopy cover	Europe	12	84	n.a.	n.a.
	Asia continent	53	n.a.	110	185
	S America	53	120	133	198

In addition to the above and below ground biomass, there is also some carbon in dead organic matter in forests. These carbon stocks are significantly higher for boreal forests, e.g. 28 t C/ha compared to tropical forests (2.1 t C/ha) (IPCC, 2006).

Furthermore, soils contain considerable amounts of carbon some of which can be lost when there is land transformation. Although there are large differences in how the land is managed, cropland that includes annual crops is generally considered to be a land use practice that tends to reduce soil carbons. Forest and grassland (perennial crops) represent land use that has a more favorable effect on soil carbon.

Deforestation is often carried out by burning biomass (after removing valuable trees for timber). This combustion is incomplete and the burning does not only release carbon dioxide into the atmosphere, but it also releases other greenhouse gases, methane and nitrous oxide of minor significance. Thus, carbon dioxide emitted when the biomass is burning and decomposing is the major GHG emission associated with deforestation (Cederberg *et al*., 2011).

In Table 4, GHG emissions estimated when different ecosystems are converted into cropland are shown based on a study by Searchinger *et al* (2008) in which the research group also included a comparison with IPCC estimates. This research modelled the potential conversion of forest/grassland into cropland as an effect of the increasing production of maize ethanol in the U.S. which indirectly would lead to a conversion of different ecosystems into cropland all over the world. The estimates assumed the loss of 25% of the carbon in top meter soils and the loss of all carbon in vegetation through burning or decomposition. As seen in Table 4, the conversion of wetlands into agricultural production would lead to high emissions due to substantial losses of soil carbon when organic soils are drained.

Table 4. GHG losses when converting different ecosystems to new cropland assuming a 25% loss of soil carbon and all carbon in vegetation (biomass) (from Searchinger et al, 2008).

Ecosystem	Estimates by Searchinger *et al* (2008) tons CO2eq/ha	IPCC (summary of different cited studies), tons CO2eq/ha
Tropical forests	553-824	604-824
Temperate forests	297-627	688-770
Tropical grassland and savannas	189-214	75-305
Temperate grassland	139-242	111-200
Wetlands	1146 (tropical moist forests in south east Asia)	748 (worldwide)

3.1.3 Effects on evapotranspiration due to Land-Use Change

Evapotranspiration (ET) is the sum of water fluxes from plant transpiration and evaporation into the atmosphere. Water movements in a plant are called transpiration and for each molecule retained, vast amounts of water enter and leave a plant. Water is lost to the atmosphere mainly in the form of vapor through stomata in the plant´s leaves. Evaporation is the movement of water into the atmosphere from soils, water bodies etc. ET requires energy which means that the process has a cooling influence.

Forests differ from cleared land in two hydrologically significant ways: they have high rates of ET and their soils allow the rapid infiltration of rain water. ET by forests is generally greater than other land covers due to low albedo and low daytime surface temperatures, high aerodynamic roughness, high leaf area and deep roots. Several studies have shown that ET is higher in tropical forests than in replacement land covers, for example pastureland. Moreover, undisturbed soils in tropical forests support rapid infiltration as opposed to deforested areas where soil can be compacted by humans, animals or vehicles (Giambelluca, 2002).

Amazonia, holding more than 40% of all remaining tropical rainforests in the world, has been the focus of many studies on hydrological dynamics. Here, the rich vegetation releases large amounts of water vapor through transpiration and together with evaporation; this equals 50-60% of the total rainfall in the region. Part of this rainfall is induced by a precipitation recycling of 25-35% (D´Almeida *et al*, 2007). Studies of Amazonia show that the spatial scale of deforestation is important in terms of its impact on the hydrological cycle. An extreme scenario, including the deforestation of the whole Amazon basin, would result in a major restructuring of land-atmospheric dynamics and lead to a severe decline of ET and on precipitation recycling, thereby weakening the hydrological cycle in Amazonia as a whole. Areas of local deforestation (<100 km^2) are too small to affect rainfall; however, run-off increases while ET decreases. Regional deforestation (100 – 10000 km^2) leads to the transformation of areas large enough to influence circulation, strengthening convection and potentially increasing rainfall (D´Almeida *et al*, 2007). In the early years of 2000, the forests of Amazonia covered about 5.4 million km^2, which represented approximately 87% of their original extent (Mahli *et al*., 2008).

3.1.4 Changes in surface albedo due to Land-Use Change

Albedo describes the amount of incoming solar shortwave radiation on a surface that is reflected back into space and is the ratio between outgoing and ingoing radiation expressed as a percentage or a value between 0 and 1 (Table 5). Light surfaces reflect much of the sun's energy and feature a high albedo factor. Dark surfaces absorb the sun's energy to warm the Earth's surface and feature a low albedo factor. The albedo factors have been derived through the use of satellites and remote sensing technology (Boeker and Grondelle, 1999; Bodikova, 2010). Accordingly, in Table 5, agricultural crops have higher albedo factors than coniferous forests, implying that changing land use from coniferous forests to agricultural crops produces a climate cooling albedo effect. Similarly, in regions with snow cover, afforestation would result in a lower surface albedo effect and hence a positive radiative forcing, resulting in a net warming despite the removal of CO_2 from the atmosphere (Pielke *et al*., 2002; Betts, 2000).

Table 5. Reflectivity values of various surfaces. Source: Budikova and Hogan (2010).

Surface	Details	Albedo
Soil	Dark and wet	0.05 -
	Light and dry	0.40
Sand		0.15 - 0.45
Grass	Long	0.16 -
	Short	0.26
Agricultural crops		0.18 - 0.25
Tundra		0.18 - 0.25
Forest	Deciduous	0.15 - 0.20
	Conferious	0.05 - 0.15
Water	Small zenith angle	0.03 - 0.10
	Large zenith angle	0.10 - 1.00
Snow	Old	0.40 -
	Fresh	0.95
Ice	Sea	0.30 - 0.45
	Glacier	0.20 - 0.40
Clouds	Thick	0.60 - 0.90
	Thin	0.30 - 0.50

The albedo effect has a significant impact on the global climate. Bala *et al*. (2006) calculate that if, theoretically speaking, the entire Southern hemisphere at mid-latitudes would be forested, that might result in a warming of 1.1°C due to the direct albedo effect. Pursuant to the fourth IPCC Assessment report (IPCC, 2007), the change in surface albedo from the expansion of agriculture since preindustrial times due to land-use change was likely to have led to a radiative forcing of −0.2±0.2W per m^2, leading to a global cooling of about −0.1°C (Kirschbaum *et al*., 2011).

3.1.5 Changes in aerosols and cloud albedo due to Land-Use Change

What are aerosols and how do they influence the climate?

An aerosol is defined as a suspension of solid or liquid particles in a gas. The word aerosol includes both the particles and the suspending gas, which is usually air. Aerosols in the atmosphere can influence the radiative forcing directly or indirectly. The direct effect is through the reflection and absorption of solar radiation into the atmosphere. Whereas most organic aerosol components scatter light and cool the Earth's atmosphere, black carbon released when forest land is burned heats the atmosphere by absorbing solar radiation. The indirect effect of aerosols occurs through modifying cloud properties. The amount of droplets in a cloud may change, thereby altering the reflection, in addition to the lifetime of cloud and precipitation patterns (Spracklen *et al*, 2008).

Atmospheric aerosols originate from either naturally occurring processes or from anthropogenic activity. Major natural aerosol sources include particles and organic compounds from forests, volcanic emissions, sea spray, and mineral dust emissions, while anthropogenic sources include emissions from industry and combustion processes (Colbeck & Lazaridis, 2010). Aerosols are also formed during the burning of biomass for energy purposes or for the clearing of land (Penner *et al*., 1992). Mineral dust from agriculturally eroded regions is another source of aerosols from land use (Colbeck & Lazaridis, 2010).

Natural aerosols from forests

Pertaining to land-use change, it is relevant to take a closer look at aerosols that are naturally produced from a forest. Vegetation can emit particles directly into the atmosphere, such as spores, fungi and leaf matter. Their contribution to the mass of aerosol may be substantial, but their climate relevance is considered relatively low (Kanakidou *et al*., 2005).

Vegetation also emits volatile organic compounds such as terpenes; these are the compounds that make the forest smell like a forest. The terpenes can in the atmosphere turn into aerosols. In a study by Spracklen *et al*. (2008), it was found that in cold climatic conditions, the snow surface albedo effect dominates the aerosol cloud albedo effects with the consequence that boreal forests warm the climate. However, in warmer zones, boreal forests may emit sufficiently large amounts of organic vapor to modify the cloud albedo effect; Spracklen *et al*. (2008) report that boreal forests can double the regional cloud condensation nuclei concentrations, implying that boreal forests can in fact in some cases cool climate.

Aerosols due to biomass burning

Aerosols also originate from biomass burning, e.g. wild fires, when clearing land, burning agricultural residues or when biomass is used for energy purposes. Biomass burning aerosols contain about 10% of black carbon by mass, which has a warming effect. However, since other primarily sulphur containing aerosols have a cooling effect, the net effect of aerosols from biomass burning is considered to be cooling.

To summarize the aerosol impact on the climate due to LUC, burning biomass to clear land will lead to emissions of sulphur containing aerosols that act to cool climate. However, forests that naturally produce organic aerosols that cool the climate will no longer will be produced if the forest was removed. Thus, in our LUC evaluation, the aerosol effect can therefore be either cooling or warming, dependent on the context.

3.1.6 Summarizing climate impacts of Land-Use Change

Afforestation could increase the availability for biomass feedstock for second generation biofuel production. Deforestation could on the other hand make more land available for annual crops. So what are the climate impacts of deforestation vs. afforestation? In a study by Bala *et al*. (2007), theoretical global-scale deforestation was simulated. The results showed a net cooling influence on the Earth's climate because the warming effect of carbon release would be outweighed by the net cooling associated with albedo and evapotranspiration changes. However, the study also indicated that there are large geographical differences. Bala *et al*. (2007) define three regions: Tropical (20°S to 20°N), temperate (20–50°in both the Northern and Southern Hemispheres), boreal (50–90° in the Northern Hemisphere). As a comparison, it can be mentioned that Sweden is located between 55°N and 69°N. Afforestation projects in the tropics would be clearly beneficial in mitigating global-scale warming. In temperate regions, the carbon and albedo effects cancel each other out on a global-mean basis and afforestation in these regions would only offer marginal benefits. Bala *et al*. (2007) conclude that due to the large albedo effects, afforestation in the boreal regions would be counterproductive.

Similar results were reached in a study by Claussen *et al*. (2001). Because of the increase in atmospheric CO_2, which outweighs the biogeophysical effects (albedo, evapotranspiration, etc), tropical deforestation tends to warm the planet. In mid- and high Northern latitudes, however, biogeophysical processes, mainly the snow-vegetation-albedo feedback through its synergism with the sea-ice-albedo feedback, defeat biogeochemical processes, thereby eventually leading to global cooling in the case of deforestation and to global warming in the case of afforestation.

In a study by Kirschbaum etl al. (2011), the afforestation of pasture in New Zealand was studied. The albedo effect of a well established forest was measured at 13% and the pasture albedo at 20 %. Following afforestation, it was found that 27 tons of carbon per hectare would need to be stored in a growing forest in order to balance the increase in radiative forcing resulting from the observed albedo change.

As pointed out by Spracklen *et al*. (2008), these types of studies ignore the impacts of atmospheric aerosol caused by natural emissions from forests or biomass burning from land clearing. Unfortunately, few studies on land-use change include aerosol impacts, making it very difficult to draw any conclusions on the climate impact of land-use change due to extended biofuel production.

To complicate matters further, there are feedback loops that need to be considered. Feedbacks are processes that amplify or dampen the effect of a forcing. For example, if changes were made to the vegetation cover (affecting greenhouse gases, aerosols and albedo effects), the climate system would respond. The ecosystem would then in turn respond to climate changes (e.g warmer climate and more precipitation could affect the type of plants, growth rate, length of growing season, etc), which in turn would effect the climate (Bonan, 2008; Carslaw *et al*., 2010; Foley *et al*., 2003) see Figure 9.

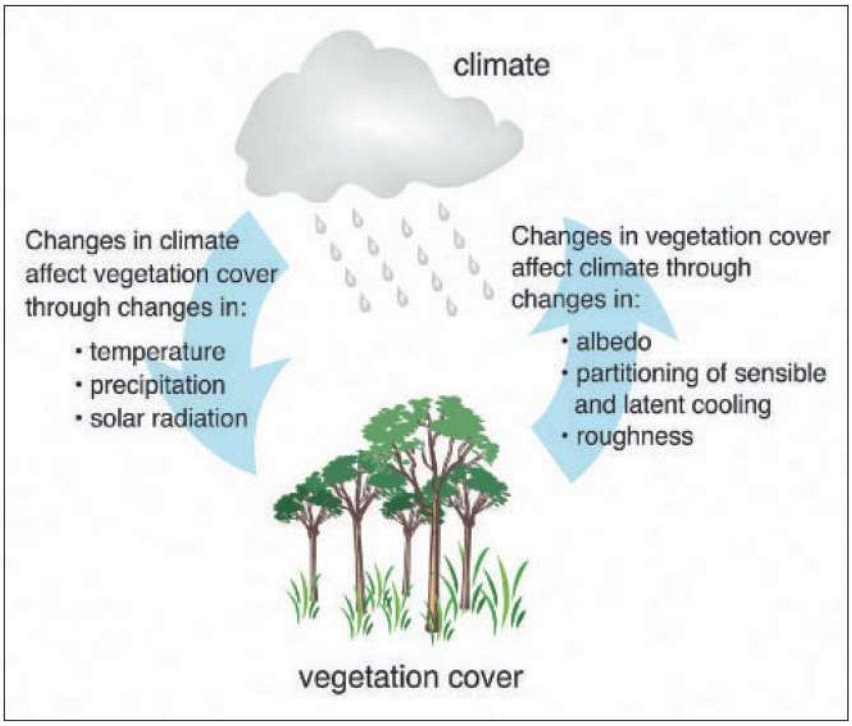

Figure 9. Biophysical feedbacks between climate and vegetation cover. Climate changes can affect vegetation cover through changes in temperature, precipitation, and net radiation. Changes in vegetation cover and surface properties can in turn affect the climate. Source: Foley et al. (2003).

3.2 QUANTIFICATION OF INDIRECT LAND-USE CHANGE

The quantification of GHG emissions due to indirect land-use changes (iLUC) are quite dissimilar from the quantifications of direct changes, as the theory in iLUC-modelling is based on economic market reactions to the increasing demand of biofuels as opposed to quantifying direct changes, which relies on natural science.

Although LUC due to bioenergy expansion was already discussed in the 1990s (see e.g. Marland & Schlamadinger, 1997; Leemans *et al*., 1996), the debate of the impact of indirect LUC on bioenergy GHG balances boomed after the publishing of two studies in 2008. Both Searchinger *et al*. (2008) and Fargione *et al*. (2008) demonstrated that iLUC could increase carbon emissions following biofuel expansion to such a level that they become more damaging than fossil fuels. These studies contributed to the EC decision to introduce an iLUC emission factor in the calculation rules for greenhouse gases (see further section 3.4).

Indirect land-use changes are not observable. A farmer in Europe starting to grow wheat for bioethanol cannot see any indirect effects and it can never be proven that a certain land use in for example Brazil is the effect of a the farmer´s change from producing wheat for food into wheat for ethanol. The linkages are complex and impossible to track down to a certain field. The only way to quantify iLUC is by using models. Two types of modelling approaches can be identified: economic equilibrium modelling and causal descriptive modelling.

3.2.1 Economic models

Economic equilibrium models are complex optimization models, which can study the entire global economy or a specific sector, such as agriculture. All equilibrium models are based on the assumption of perfect markets and that equilibrium is reached when demand equals supply in the economy studied. The different models optimize different benefits; in a partial model of agriculture, the farmers' profits might be optimized. In general equilibrium models, it may be corporate profits that are maximized (Widell, 2009).

There are several different economic models, which have been developed and used by researchers for many years. Economic models typically assess changes associated with the implementation of a policy, such as a biofuel policy. The land-use change is often calculated as the difference between a scenario with and a scenario without the implementation of a policy. Most models are complex, not transparent and can only be run by researchers who have in-depth knowledge of the model.

3.2.2 Casual descriptive models

Several “simplified” alternatives to the economic models have been developed, with varying approaches. The common feature is that they try to use descriptive methods rather than complex optimization models. One model, for example, lets reference expert groups describe likely scenarios of market reaction to the increased demand of biofuel (Bauen *et al*., 2010). Others use statistics of past land-use changes to predict future land-use changes (for example Fritsche *et al*., 2010; Tipper *et al*., 2009).

3.2.3 iLUC factors

Most economic models gives results as the number of hectares of additional land needed for the introduction of a biofuels policy, and regions involved. But they do not give answers to exactly where the change will take place and what type of land that will be affected. Therefore, additional assumptions on which land is going to change and the amount of GHG emissions that are associated with the change are needed. For this purpose, other types of biophysical models are used. The GHG emissions associated with indirect land-use change is often referred to as iLUC-factors, expressed as the CO_2-equivalents per MJ biofuel.

When land use changes, there can be a large initial loss of carbon stocks followed by a number of years before a new equilibrium is reached. Usually, the total amount of carbon lost is divided equally over a certain number of years, so that a crop grown right after a conversion and several years after the conversion will be penalized with the same emission factor. Often, the emissions are divided over 20, 30 or 50 years.

In Figure 10, the resulting GHG emissions from different modelling efforts are presented. A couple of studies show impacts below zero. This is because they assume that by-products from the biofuel production replace high-emitting alternatives, such as soy grown in previous rain forest.

As a comparison to the biofuel’s GHG emissions, the tailpipe emissions of 1 MJ petrol is approximately 72 g CO2eq and including direct and indirect emissions from production and distribution leads to higher emissions. The value of petrol used for comparison in the EU Renewable Energy Directive is 83.4 g CO2eq per MJ. According to Liska and Perrin (2009), petrol can cause up to 195 g CO2eq per MJ if indirect effects, such as military efforts in oil producing countries, are included.

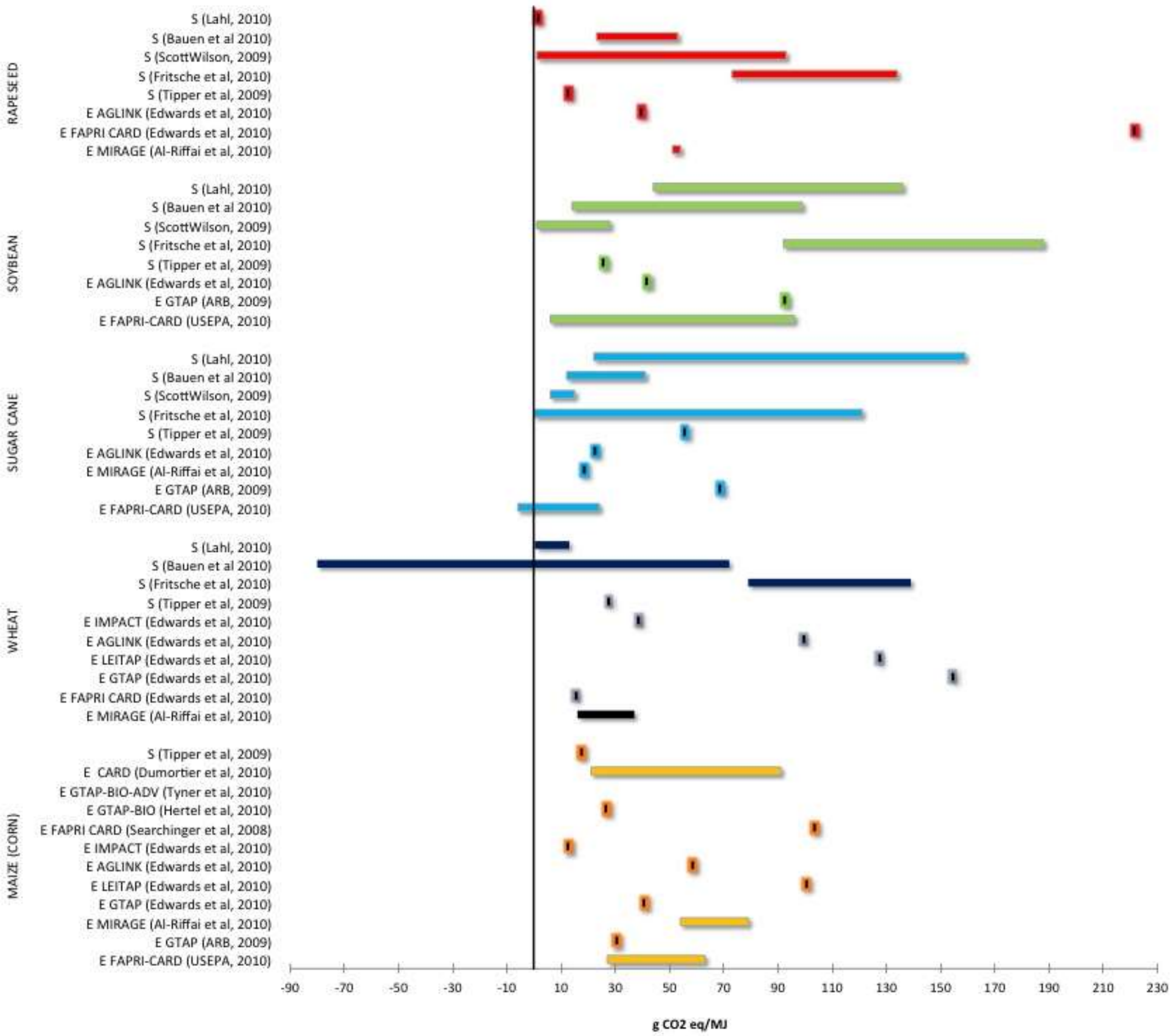

Figure 10. Model results of annualized GHG emissions (g CO2 equiv./MJ) due to LUC. Notes: Some studies show results as intervals (illustrated with lines), others as specific values (illustrated with dots). The emissions are allocated over a period of 20 years. 'E' stands for equilibrium models, 'S' simplified models. The large variation in results from the different studies clearly illustrates the high uncertainty associated with modelling efforts. The image has previously been published by Di Lucia et al. (2012).

3.2.4 Why such different results?

Model assessments of LUC display large variations in results (Figure 10). There are many reasons for the large divergence found.

The choice of model is of course important: if an economic or descriptive model were used. It is also important to consider that all models are projections of future changes, and that it would be remarkable if all models showed the same results.

Most models are incapable of distinguishing between dLUC and iLUC. The economic models, for example, give the results as the land-use change between a scenario with, and a scenario without, the implementation of a biofuel policy. The total LUC on a global level broken down into smaller units of different biofuels is reported. Some of the casual descriptive models have the same top-down approach, assessing total LUC. However, most studies report the results as iLUC even though it is not always obvious if dLUC or iLUC is included.

For the economic models specifically, there are several reasons for the varying results directly connected to the modelling (Cornelissen *et al*., 2009; DGEnergy, 2010; Khanna and Crago, 2011; Nassar *et al*., 2011; O´Connor, 2011; Prins *et al*., 2010; Yeh and Witcover, 2010):

- The economic models are originally developed for purposes other than assessing iLUC. They represent different world views and, therefore, different assumptions for the development of oil prices, food prices, etc.
- Some economic models study entire economies (general equilibrium models), while others only study specific sectors of the market (partial equilibrium models).
- Different policies and different end points are studied. The ratio between biodiesel and ethanol that is produced to fulfill a policy goal can differ and in some models, second generation fuels are also included.
- The geographical resolution differs. Some models study the crop trade in every country in detail, while others aggregate larger areas.
- The commodity level resolution varies. While some models study “cereals”, others can differentiate between wheat, oat, etc. Some models also allow for forest products to be used as biofuel raw material.
- Assumptions on harvest levels and raw material needs per MJ biofuel may differ.
- Assumptions on the amount of by-products and how these are valued may differ. Some studies do not account for by-products at all.
- The assumptions on how the demand for different commodities depends on, for example, the price of commodities (the so-called elasticity factors) is of great importance and varies between the models. Also how the import/export relationship is affected by price changes is an important assumption.
- Some models allow the trade of biofuels, while others do not.
- How the expansion of agricultural land is modelled differ. Some allow expansion on pasture, others in forest and yet others both. The prices for land and costs of converting land are other important assumptions for how land use is modelled.
- Several models cannot distinguish between deforestation and avoided afforestation.
- Nitrous oxide emissions are sometimes included, on other occasions not.

Efforts have been made to deal with the modelling uncertainty. In a study by IFPRI (2011), the uncertainty range of the economic model MIRAGE was studied (Figure 11). The uncertainty range was assessed using the Monte Carlo simulation, in which a large number of simulations are conducted, randomly selecting parameter values (each parameter is attributed to an individual uncertainty range). This, however, only shows the uncertainty distribution of one specific model.

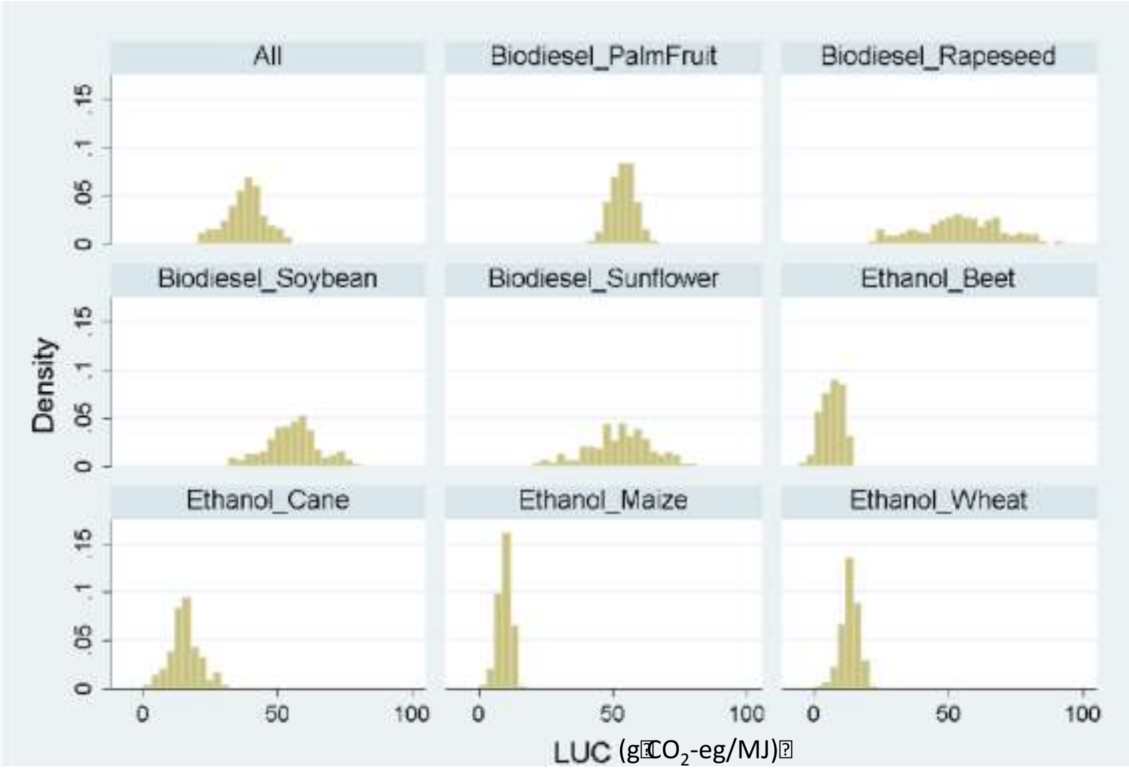

Figure 11. Results of Monte Carlo simulation based on the economic equilibrium model MIRAGE for a number of different biofuels (IFPRI, 2011).The LUC is expressed as g CO2-eq/MJ biofuel, the density representing the frequency (based on random selection of parameter values). The results show the probability interval for LUC emissions. For example, 95% of the simulation results for wheat ethanol is within the range of 8 and 18 g CO2-eq/MJ, with a median value of 14 g CO2-eq/MJ.

About 99% of the Monte Carlo simulations in the IFPRI study showed positive LUC-emissions (on the average 40 g CO_2eq/MJ biofuel). This means, according to the same study, that about half of the gains from switching from fossil fuels to biofuels vanish even without taking into account the actual processing steps of the biofuels (IFPRI 2011).

The report by IFPRI (2011) concludes that land-use related emissions eliminate more than two-thirds of the direct emission savings. The same report also assesses different feedstock options in relation to resulting emissions. Sugar beets for example, have the lowest land-use emissions coefficient and sugar cane the highest coefficient. However, only considering the net emissions, sugar cane seems to be the best feedstock. Effective processing technologies and resulting co-products make sugar cane the best option.

Sunflower oil seems to be the best option when it comes to vegetable oils compared to soybean, which has the highest LUC emissions. In general, the differences between emission coefficients among vegetable oils are profoundly small. Sunflower and palm oil are, according to the models and assumptions used, the only biodiesel feedstock that results in small net emissions savings for biodiesel – roughly 6% of the fossil fuel reference.

In a study by Plevin *et al* (2010), the uncertainty in iLUC emissions from maize ethanol was studied using a Monte Carlo simulation of a simplified model. The results showed that the 95% interval was 25 - 150 g CO_2-equivalents per MJ maize ethanol, with a median value of 65 g CO_2-equivalents per MJ.

3.2.5 Magnitude of carbon stock changes - another source of uncertainty

The means by which indirect land-use changes produce a greenhouse gas impact is through changes in carbon stocks, i.e. the difference between carbon stored in biomass and soils before the land transformation and carbon stocks in vegetation and soils after the transformation. Calculations of carbon stock changes are crucial for the results of iLUC-studies, but an extensive EU review shows significant differences between the various models and surprisingly few discussions of methodological and data issues used in the studies. Important differences between studies include (DGEnergy, 2010):

- *The carbon stock values that are attributed to different land uses in the original ecosystem.* This is an important parameter. Some of the disparities between studies may be attributable to differences in the exact parcels of land each value is meant to cover. For example, does the land transformation take place in the Amazon rainforest biome or in the Cerrado biome in Brazil? Obviously, the original vegetation in Amazon forests has substantially higher carbon stocks than in the Cerrado biome. Also, there are spatial variations in carbon stocks within these two biomes (Batlle-Bayer *et al*. 2010; Cederberg *et al*., 2011).

- *The proportion of carbon stocks that is lost during land transformation*. In the case of the conversion of forest land to cropland, studies featuring high estimates of carbon-stock changes assume that all above-ground carbon and 25% of soil carbon are lost, while studies featuring low estimates assume that only 75% of carbon in vegetation is lost.

- *The carbon stock that is attributed to vegetation on cropland (i.e., to crops).* This can vary from zero to 5 ton CO_2 per hectare (for most crops) and 17 ton CO_2 per hectare for sugar cane.

- *How foregone sequestration is treated,* i.e. the take-up of carbon from the atmosphere by forests/vegetation that is lost after the land transformation. Some studies take this into account, others do not.

- *The values attributed to foregone sequestration.* In the studies reviewed the highest number is around 5 ton CO_2 per hectare and year of forest converted to cropland (EU included in some studies) while 0-2 ton CO_2 per hectare and year is applied for North America (in some studies). None of the studies reviewed by the DG Energy take into account foregone sequestration from converted grassland. Annual carbon sequestration rates of between 0.7 and 1.8 ton CO_2 per hectare and year have been measured and calculated for continental European grasslands (Sousanna *et al*., 2010). As an average for the entire European continent, Schultze *et al*. (2009) estimate the average carbon sink on the European grasslands at around 2 ton CO_2 per hectare and year. Recent studies of non-fertilized grazed grassland in the U.S. indicate a yearly sequestration of 1.4 ton CO_2 per hectare and year (Liebig *et al*., 2010).

Of the differences listed above in how carbon stocks are handled in studies of iLUC, estimates of carbon stock values in the original vegetation are the most important (including biomass above as well as below ground). According to DG Energy (2010), even when the literature reference of carbon-stock values is supposed to be similar, there are still large differences between studies. Taking IPCC-data as examples, which are often used in studies of iLUC, a model using this data states that carbon stock losses are 322 tCO_2/ha when forest is converted to non-forest use, while another model using the same source assumes losses of 299-627 ton CO_2 per hectare (temperate) and 553-824 per hectare (tropical) for forest to cropland conversion (DG Energy, 2010).

3.2.6 Knowledge gaps

There are different kinds of uncertainties in the assessment of land-use changes induced by an expansion of biofuel production, as well as of its potential effects on GHG balances and GWP. One category includes inherent uncertainties, which persist independently of knowledge. Examples are the quantification of indirect land-use changes which is not observable. An increased knowledge of modelling iLUC, using either economic models or casual descriptive models, will improve our understanding of the pros and cons of the various models, but never lead to a conclusive answer of how an additional biofuel crop cultivation in, for example, Europe would affect land use in South America. Another example of inherent uncertainties relates to the potential effects of iLUC on biogenic GHG balances. Since iLUC is not observable, neither are its potential effects on the biogenic GHG balance. A better knowledge of the most crucial parameters in the quantification of iLUC and its potential GHG consequences, may, however, be useful in our efforts to reduce the risk of the negative effects of an expanded biofuel production. Such increased knowledge could be valuable in designing strategies, policies, and political tools minimizing the risks of negative iLUC.

Contrary to iLUC, dLUC caused by an expansion in the biofuel crop cultivation is detectable. However, the quantification of its consequential changes on biogenic GHG balances often suffers from high uncertainties due to the lack of reliable data. Here, increased knowledge about the historical cultivation systems, soil management on the actual site, the soil carbon content and its past trend, in addition to other specific local conditions, will reduce the uncertainties associated with quantifying carbon stock changes. An increased knowledge of the specific local conditions including biological, physical, and chemical soil properties will also improve the estimations of changes in the potential risks of nitrous oxide emissions. However, due to the complexity in the formation of nitrous oxide induced by a large multiplicity of parameters, such estimates made by more site specific models will still include inherent uncertainties. An improved knowledge of the most critical factors is, nevertheless, valuable when developing general strategies for reducing the risk of biogenic nitrous oxide formation.

Land-use change will affect the climate not only by the change in GHG emissions, but also by changes in albedo and evapotranspiration when, for example, forest land is converted to arable land. Today, these parameters are not considered in land-use change models linked to increased biofuel production. However, according to the IPCC (2007), the climate effects of direct aerosol, aerosol-cloud interactions, and evapotranspiration contain considerable uncertainties in contrast to the relatively high confidence in quantifying the climate effects of GHG emissions. Further, there are feedback loops between climate and vegetation, which are poorly understood. The shape of the landscape (surface roughness) can also play a role. Taking into account all these parameters will further increase the uncertainty. Nevertheless, an improved knowledge of LUC and its impact on the climate is a prerequisite for the formation of general and effective strategies preventing significant negative effects as a result of an expanded biofuel production.

Current modelling of land-use change and related effects on the potential for global warming often comprise different time and spatial perspectives. The knowledge about the importance of various perspectives of the resulting changes in GWP has increased during recent years, but there is still a knowledge gap considering these aspects. For example, the changes in the carbon stock owing to dLUC are usually evenly divided over a certain period of years in the life cycle assessment of biofuels. This is a simplified approach sometimes leading to questionable interpretations of the resulting GHG performance of the biofuel system studied. An increased knowledge about the site-specific conditions and historical soil management opens up for an improved and more dynamic approach leading to a more consistent evaluation of the time aspect. However, since this depends on the selected reference year which often differs between different studies, there will still be room for various interpretations of the consequential changes in the GWP.

The spatial perspective may also differ in different models of the climate effects of LUC. Depending on whether the model considers an individual cultivation, plantation or forest stand or applies a landscape perspective including a large variety of cultivations, plantations or forest stands, the effects of LUC on the GWP will be significantly different. Thus, the issue of the spatial perspective in biofuel modelling is not primarily a matter of increased knowledge and improved input data, but rather a matter of the relevance of the applicable perspectives.

As relevant knowledge and insights increase, the overall methodology in the assessment of the GHG performance of biofuels is changing over time. However, there will always exist a certain degree of flexibility and freedom in the choice of approaches and assumptions in the methodologies developed. One example relates to the definition of systems boundaries. A sufficient knowledge about the consequences of different systems boundaries on the results is a prerequisite for making relevant interpretations and rational conclusions. The systems boundaries could include a specific biofuel-production system where, for example, by-products and their indirect GHG effects are assessed. In previous models, the handling of by-products is not scientifically consistent where, for example, the LUC caused by the biofuel (often negative) and by the by-products (often positive), respectively, is not equally assessed. Thus, there is currently a knowledge gap among some modellers regarding the scientific reliability in the methodology of systems studies, which leads to inaccurate results.

The systems boundaries could also reflect a much broader perspective, such as the global agricultural system. In this perspective, the issue of direct and indirect land-use changes and their mutual relationship, could be questioned. Encompassing such a broad system perspective, one can argue that the LUC is often double counted. For example, if an expanded biofuel crop production were to take place on excess arable land leading to dLUC, these production systems should not be burdened by any iLUC effects, which is often the case in current models. Furthermore, one farmer’s iLUC is another farmer’s dLUC; thus, including the entire global agriculture systems and all food, feed and biofuel production, from a theoretical perspective there is no iLUC. The knowledge about these relationships is limited today and needs to be improved. Another issue of relevance is the assessment of the direct and indirect GHG effects of the current fossil fuel system used as a reference. Applying a broad system perspective, taking into account all the various indirect effects caused by current fossil-based systems, the GHG performance of comparable petrol and diesel fuels could be significantly worse than current fossil reference systems show. Thus, another knowledge gap is identified.

3.3 EFFECTS ON FOOD PRICES, FOOD SECURITY, POVERTY AND LAND RIGHTS

Against a backdrop of nearly three decades of declining or stable food prices, the world saw a sudden and sharp increase in the price of basic agricultural commodities during the 2007-2008 period. Between January, 2007 and June, 2008, real world market prices of wheat and corn increased by about 60%, the price of rice increased by 124%, fats and oils increased by 91%, and beef prices by 26% (World Bank 2012), see Figure 12. Taking the world by surprise, the 2007-2008 food price crisis sparked both public upheavals across the world and an intense – and sometimes heated – debate on the causes and consequences of the price increases.

At the center of the spotlight was the diversion of agricultural land in developed countries – mainly the EU and US – from the production of food to biofuel feedstock production. International organizations, such as the World Bank and the International Monetary Fund (IMF), claimed that the lion's share of the price increases was attributable to increased demands for lands to cultivate biofuels (Ciaian and Kancs 2011), with the U.N. special rapporteur on the right to food, Jean Ziegler, going as far as to describe the diversion of arable land to the production of biofuel feedstock as a "crime against humanity", calling for a five-year ban on biofuel production. The U.S. and EU on the other hand tried to downplay the role of biofuels, with the European Commission arguing that its modest use of cereal for the production of ethanol was a "drop in the ocean","not something to shake the markets" (Ciaian and Kancs 2011).

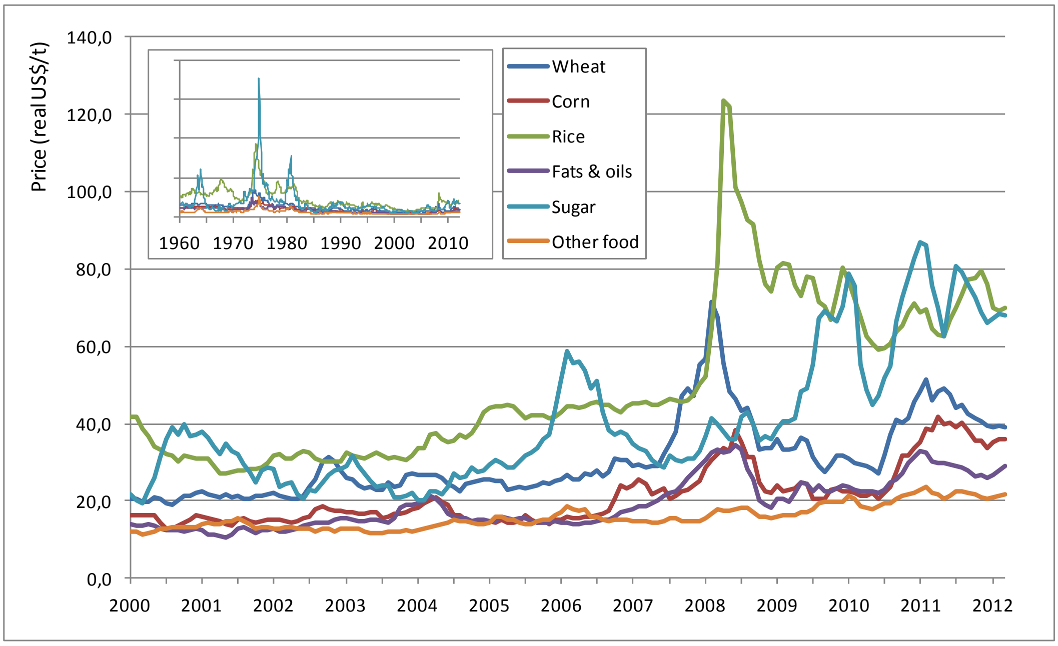

Figure 12. Monthly world market prices for major agricultural and food commodities in the January 2000 to March 2012 period (and in the inset from 1960-2012). Data from the World Bank (2012).

While world prices dropped back to lower levels after the 2008 spike, they dropped back to levels higher than those prevailing prior to the crisis; 2011 again saw increases in agricultural commodity prices. Thus, although the tone of the debate has toned down, the effect that an increased demand for biofuels has on food prices and, consequently, on poverty and malnutrition is still ongoing.

Although agricultural markets have long been connected to energy markets through the use of inputs (fossil energy and fertilizers), several studies present empirical evidence of an increased integration between energy and agricultural markets due to the increased use of biofuels (FAO 2008; Tyner and Taheripour 2008; Gohin and Chantret 2010; Headey *et al*. 2010; Hertel 2010; Tyner 2010; Ciaian and Kancs 2011). It is important to recognize that the basic mechanism through which this new linkage is established is the competition for arable land; if agricultural land resources were unlimited, the increased demand for biofuels would not affect food prices. In the biofuels debate, it is sometimes argued that the problem is that we use food (e.g., corn or wheat) to produce biofuels, not non-food feedstocks such as cellulose. However, as long as the production of feedstock requires agricultural land, higher demand for biofuels will tend to drive up food prices, whether the actual feedstock can be eaten or not (though, as noted below, if the yield of second generation feedstocks were higher, this would lessen the competition for land and hence the effect on food prices).

While the basic causality from increased biofuel use to welfare effects for the world's poor is relatively straightforward. – i.e., higher demand for biofuels leads to agricultural land being diverted to produce biofuel feedstock, leading to lower food production and higher food prices, in turn affecting malnutrition and poverty – the are many real world complexities involved in tracing each step in this chain (Headey and Fan 2008). First, the effect of an increased demand for biofuels in one country on agricultural commodity prices will depend on the responsiveness of supply and demand, such as the possibility to increase the cropland area, increase agricultural yields or substitute feedstock crops in consumption (Naylor *et al*. 2008). Second, higher domestic prices in biofuel producing countries will not perfectly translate to increases in world market prices or in turn to increases in prices facing poor consumers in developing countries due to trade distortions, transactions and transport costs and exchange rate movements (Conforti 2004). Third, the welfare effect of higher food prices in developing countries will depend on, *inter alia*, the positions of the country and the individual household as net buyers or sellers of food, countervailing policy responses, and the existence of social safety nets (Headey and Fan 2008). Fourth, the welfare effects of higher food prices may be swamped by other societal changes in developing countries, most notably increasing income levels (Headey 2011).

Below, we review each of these issues, beginning with available estimates of the extent to which the increased demand for land for biofuel production contributes to recent and future food price changes, including the issue of price transmission from world markets to local markets in developing countries. Thereafter, we review the literature on how these food price increases may have contributed to changes in the prevalence of food insecurity and hunger in developing countries and the extent to which the price increases have fuelled the global rush for farmland in developing countries and the resulting effects on land rights and livelihoods of the rural poor. Finally, the main findings are summarized and discussed.

3.3.1 The impact on food prices of increased demand for land for biofuel production

Numerous studies have tried to assess the impact of the present and future demand for land for biofuel feedstock production on food prices. While some studies qualitatively discuss the role of increaseed biofuel production for the 2007-2008 food price hike (e.g., Abbott *et al*. 2008; FAO 2008; Headey and Fan 2008; Mitchell 2008; Trostle 2008; Abbott *et al*. 2009; Piesse and Thritle 2009; Headey *et al*. 2010; Abbott *et al*. 2011), others try to quantify the magnitude of the effect of biofuel demand on agricultural commodity markets. The results of a number of the latter studies are summarized in

Table **6**.

The general conclusion emerging from the studies analyzing recent food prices changes is that the demand for land for biofuel feedstock production has been a major, though not the sole, contributor to prices increases (other important factors identified are declining stock-to-utilization ratios, the depreciation of the dollar, rising oil prices, and – in the case of rice – export policies, though the relative importance of these drivers are disputed; the effect of other factors such as financial speculation, poor harvests, and increased demand for agricultural commodities from growing economies in China and India are generally downplayed) (Abbott *et al*. 2008; Headey and Fan 2008; Mitchell 2008; Trostle 2008; Piesse and Thritle 2009). Quantitative estimates generally suggest that about 30-50% of the increase in cereal prices in the 2007-2008 price spike was due to the increased demand for biofuels (Rajagopal *et al*. 2007; Birur *et al*. 2008; Rosegrant 2008; CBO 2009; Rajagopal *et al*. 2009; Roberts and Schlenker 2009; Babcock 2011; Hausman *et al*. 2012).

However, as seen in Table 6, the estimated effect on recent and future food prices vary considerably between studies. Part of the reason for diverging results is the fact that different studies assume varying increases in the demand for biofuels (e.g., some studies estimate the effect of only U.S. biofuels policy, others U.S. and EU policies, and yet others increased demands from a larger group of countries). This difference can be accounted for by dividing the estimated price increase for a given biofuel feedstock by the size of the demand shock, calculating so-called shock multipliers that tells us how much the price of a given commodity would increase if the biofuel demand were increased by a given amount (see e.g., Fabiosa *et al*. 2010; Timilsina *et al*. 2010), (see Table 6). Thus, the multiplier produces a more general estimate of how sensitive food prices are to the increased demand for land for biofuels. As seen in

Table **6**, there are still large disparities between studies; for corn ethanol, multipliers range between -0.96 %/EJ and 1.58 %/EJ (mean 0.20 %/EJ, standard deviation 0.35 %/EJ) and for biodiesel from oilseeds, multipliers range between 0.01 %/EJ and 0.82 %/EJ (mean 0.29 %/EJ, standard deviation 0.25 %/EJ).

These large differences in how sensitive food prices are to increases in feedstock land demand stem from to the large differences in methodology and assumptions used in different studies, making it difficult to compare results between studies (Mitchell 2008; see also the discussion on model estimates of LUC above). Studies analyzing recent food price increases assume a range of different approaches, from relatively simple calculations using demand and supply elasticities for agricultural commodities, to conceptual partial economic models, regression analyses and detailed economic optimization models (partial and general equilibrium models), while studies generating scenarios for the future generally adopt the latter approach.

Despite the difficulty of comparing the results of different studies, a number of robust conclusions emerge from the literature on the effect of increased demand for biofuels on food prices. First, increases in prices can be expected to be higher for the crops used directly as biofuel feedstock in the regions where the increased demand for biofuel occurs. These results are evident in studies that report price changes for multiple crops (e.g., Babcock 2011; Chen *et al*. 2011) and regions (Birur *et al*. 2008; Kretschmer *et al*. 2009), as well as across studies – average price multipliers are higher for both corn and oilseeds in the regions where the greatest increase in biofuel demand occurs, the EU and U.S.[5]

The reason price increases will tend to be higher for the crops used as biofuel feedstock is the imperfect substitutability both on the supply and demand side. Because not all crops can be grown on all agricultural land, an increased demand for one crop may have only limited impact on the supply of other crops grown on land for which the first crop is not suitable. A case in point is rice production in the U.S., which according to Babcock (2011) faces less competition for land than other crops and, therefore, experiences only minor increases in price due to increases in demand for, e.g., corn ethanol. Similarly, on the demand side, the fact that different agricultural commodities are not perfect substitutes for the consumption implies that price effects will be highest for those crops directly experiencing increases in demand.

Moreover, although these results are not presented in

Table **6**, studies that also report price changes in other food commodities, such as meat, dairy, eggs, and processed foods, generally find the price of these commodities increase less than those of basic agricultural commodities (e.g., Birur *et al*. 2008; Hayes *et al*. 2009; Timilsina *et al*. 2010; Chen *et al*. 2011). The reason is that prices for these goods are not solely determined by prices of basic agricultural commodities, but also by other factors such as wages, energy, transport, and storage.

In addition, some of these inputs may experience price drops in scenarios where the use of biofuels increase; mandated use of biofuels will lower the demand for gasoline and diesel, thereby reducing the price of these fuels (e.g., Rajagopal *et al*. 2009; Timilsina *et al*. 2010; Chen *et al*. 2011), and the increased production of some biofuels (e.g., corn ethanol and rapeseed biodiesel) produce by-products that can be used as animal feed, lowering the price of this input (Schmidhuber 2007; OECD 2008; Fischer *et al*. 2009). The effect of these developments on mitigating the adverse impact on final consumers can, however, be expected to be strongest in developed countries where farm-gate prices of agricultural commodities only constitute on the average 20-35% of the final food retail price; in low-income countries where basic foodstuffs make up a larger share of the food basket, the effect of rising food prices on consumers can be expected to be higher (Dewbre *et al*. 2008).

Second, models estimating price increases using partial equilibrium (PE) models generally find larger impacts than general equilibrium (CGE) models[6]. The reason is the manner in which a demand shock propagates through the different models; in global CGE models, patterns of production, consumption, and trade may change substantially in response to a shock, leading to minor effects on prices, while in PE models, where changes in production and consumption cannot take place across different sectors of the economy, a larger part of the adjustment comes through price changes (Wiggins *et al*. 2008).

[5] The average multipliers for corn are 0.26 %/EJ for EU and US prices and 0.14 %/EJ for world prices, while average multipliers for oilseeds are 0.34 %/EJ for EU and US prices and 0.27 %/EJ for world prices. Differences in means are statistically different at the 16% and 30% level in a one-tail t-test for corn and oilseeds, respectively.
[6] The average multipliers for corn are 0.23 %/EJ for PE models and 0.07 %/EJ for CGE models, while average multipliers for oilseeds are 0.37 %/EJ for PE models and 0.20 %/EJ for CGE models. Differences in means are statistically different at the 3% and 8% level in a one-tail t-test for corn and oilseeds, respectively.

In that sense, PE models may better depict effects of increases in the demand for land for biofuels in the short run, where different constraints make market adjustment costly or impossible, whereas CGE models better reflect the medium to long term response of agricultural prices to increased demand (Mitchell 2008; Wiggins *et al*. 2008). This difference between short-run and long-run price responses to biofuel demand is also evident in the studies summarized in

Table **6**, where the average price multiplier for corn is 0.30 %/EJ in studies of recent food price increases and 0.13 %/EJ in studies exploring scenarios for the future[7].

Third, price increases are higher in the case when there is no market penetration of second generation biofuels. Again, this is evident both within studies (Msangi *et al*. 2007; OECD 2008; Hayes *et al*. 2009; Chakravorty *et al*. 2011; Chen *et al*. 2011), as well as in comparisons across studies; the average price multipliers for studies that allow for some utilization of second generation biofuels are 0.09 %/EJ and 018 %/EJ for corn and oilseeds, respectively, while multipliers for studies that only see an expansion of first generation biofuels average 0.24 %/EJ and 0.33 %/EJ, for corn and oilseeds, respectively[8]. As discussed above, this follows directly from the fact that the production of second generation biofuels requires less land area for a given demand of biofuels.

In addition to the above, we should intuitively expect estimated price increases to be higher in models that do not allow cropland areas to expand, or only expand at the expense of other managed lands (Chakravorty *et al*. 2011), see

Table **6**. A comparison across studies does not support this proposition, however, although this may be due to other model differences and the small sample. The only study to test this formally (Popp *et al*. 2011) finds that prohibiting the expansion of crop-land into current intact and frontier forests drastically raises food prices compared to the case when deforestation is allowed, especially in vulnerable regions such as Sub-Saharan Africa, South Asia, and Latin America (though the magnitude of the effect is somewhat surprising, given that bio-energy prices hardly differ between the two scenarios). These results suggest that there might be a trade-off between mitigating the environmental and socio-economic detriments of increased demand of land for biofuel production, with small land-use changes leading to higher food price changes, and vice versa.

Finally, as noted above, even if increased biofuel demand drive up world market prices, these increases may not be fully transmitted to local markets in low-income countries due to, e.g., trade barriers, countervailing policies, or transaction and transportation costs. Several studies have analyzed the level of price transmission from world markets to low and middle-income countries statistically (Conforti 2004; Minot 2011; Brown *et al*. 2012). Although results show a wide variation of market integration between and within countries and for different commodities, two general patterns emerge: (1) agricultural markets in Latin America and Asia are generally well integrated in world markets, while African markets are less integrated; (2) market integration is higher for

[7] The difference is significant at the 12% level in a one-tail t-test. The average price multiplier for oilseeds is actually higher for the analyses of historic price increases, than for scenario studies. However, this is most likely due to the fact that there is only one study (Birur et al, 2008) that tries to estimate the contribution of increased biodiesel demand in the EU to recent food price increases, and this study uses a CGE model.
[8] The difference between studies including second generation biofuels and those that do not is significant at the 5% and 7% level in a one-tail t-test for corn and oilseeds, respectively.

widely traded cereals (i.e., wheat, maize, rice), than for locally produced staples (e.g., cassava, plantains, beans), oilseeds, and livestock.

Even though African agricultural markets historically have exhibited imperfect integration, the study by Minot (2011) shows that the price transmission from world market prices during the 2007-2008 food crisis was high also in many parts of Africa; the average price transmission was 42 percent in West Africa, 97 percent in East Africa, and 107 percent in Southern Africa. This divergence from the historical record may partly be explained by the concurrent increase in oil prices (raising transport costs, especially in landlocked countries), weather shocks, and the magnitude of the changes in world prices (Minot 2011). With reductions in trade distortions and improvements in infrastructure, we should also expect market integration and price transmission to increase over time.

3.3.2 The effect of higher food prices on food security and poverty

In analyzing the effect of increasing food prices on poverty, a couple of distinctions are important to make. First, there is a difference between vulnerability to higher food prices in a macro and microeconomic sense; i.e., countries with high poverty and malnutrition rates, and subsequently a large share of households that may suffer from price increases, may still on the aggregate level be less sensitive if price transmission were low or if they were net exporters of food. Second, in evaluating the effect of recent food price increases, there is a difference between the poverty and hunger impacts of rising food prices alone – which is what most studies have analyzed – and the net impact accounting for concurrent changes in, e.g., income levels and other commodity prices.

Starting with the macro-economic effect of food price inflation on poor countries, most low income countries are net importers of food (Ng and Aksoy 2008), have until recently witnessed deteriorating terms of trade in food (Schmidhuber 2007), and would most likely stand to lose from further food price increases. Although numbers have dropped in recent years, the FAO still lists 66 countries as low-income and food-deficit nations (i.e., net-importers of food), the majority of which are in Africa, see Figure 13. Consistent with this, countries included in the modeling exercises previously discussed that reported welfare (i.e., GDP) effects of increased biofuel use reported negative consequences for low and middle-income countries (Wiggins *et al*. 2008; Cororaton *et al*. 2010; Timilsina *et al*. 2010; Chakravorty *et al*. 2011; de Hoyos and Medvedev 2011), with the exception of prospective biofuel exporters, such as Brazil and Thailand (Cororaton *et al*. 2010; Timilsina *et al*. 2010).

Figure 13. Countries shaded in dark are defined as low-income, food deficit (LIFC) nations. Source: FAO (2012).

Just as most low-income countries are net importers of food, household surveys from these countries consistently find that the major share of the population are net consumers of food and would lose from higher food prices, at least in the short run (Ivanic and Martin 2008; Wodon *et al*. 2008; Zezza *et al*. 2009; Jayne *et al*. 2010; Bryngelsson *et al*. 2012). While this result is not surprising for urban areas, where the share of net buyers often exceed 90-95 percent, the fact that net sellers constitute less than half of the rural population in nearly all developing countries studied, and less than a third in most of them, may be more unexpected (Bryngelsson *et al*. 2012).

Several studies have used the data from household surveys to simulate the impact of recent food price increases on household disposable income and, consequently, on poverty rates and malnutrition. The results present a relatively consistent geographical pattern of effects, with the largest negative welfare impacts in South Asia and Sub-Saharan Africa (Cororaton *et al*. 2010; Tiwari and Zaman 2010; de Hoyos and Medvedev 2011). Ivanic and Martin (2008), using household data from nine developing countries, estimate that world market food price increases between 2005 and the first quarter of 2008 would increase poverty (measured at the $1/day poverty line) by slightly over 100 million people across all low-income countries. Wodon *et al*. (2008), using household survey data from twelve Central and West African countries, find that a 50 percent increase in the basket price of selected food items would increase the share of the population in these countries living in poverty by about 3.5 percent. Extrapolating this result to the rest of Sub-Saharan Africa implies 30 million people falling into poverty as a result of the food crisis.

Using a much larger dataset covering 73 countries, de Hoyos and Medvedev (2011) estimate that food price increases until 2007 might have pushed up poverty (measured at the $1.25/day poverty line) by about 150 million.

Tiwari and Zaman (2010) estimate that the spike in food prices during the 2007-2008 period increased the prevalence of undernourishment by 86-113 million across the developing world, using a model based on the relationship between income, income distribution, and food consumption from 74 countries,. Using a similar methodology and data from 70 low-income countries, USDA (Rosen *et al*. 2008; Shapouri *et al*. 2009) found that the number of food insecure individuals (i.e., people who are not able to meet nutritional requirements) in the countries analyzed increased by 133 million in 2007 and by another 80 million in 2008.

The above studies caution that the estimated poverty and malnutrition effects are merely indicative and should be treated with great caution. Headey (2011) lists a number of issues that are problematic with these estimates. Primarily, the empirical basis for the simulation exercises is in many cases weak. This weakness applies to both the number of countries sampled – which even in the studies that have a broader coverage excludes large middle-income countries such as China, Brazil, and Mexico – as well as model assumptions (e.g., price elasticity of calorie intake).

Furthermore, no studies except those by de Hoyos and Medvedev (2011), apply actual domestic data on price increases in developing countries, but assume an average transmission of world price increases to domestic markets; nor do they account for any policies that were introduced to cushion the impacts of food price inflation. While, as noted above, price transmission from world to domestic markets were on the average high during the 2007-2008 price spike, the high variation of price transmission among countries, together with large variability in the vulnerability to food price changes, implies that impact estimates might be affected.

A means of overcoming many of the empirical problems inherent in the simulation studies would be to instead rely on reported impacts on food security and hunger. Brinkman *et al*. (2010) summarize the results of the 2008 Food Security Assessment of the World Food Program covering 19 developing countries and find widespread negative effects on food security, both in terms of quality and quantity of food consumed. Headey (2011) utilizes responses from the Gallup World Poll (GWP) that has been sampling households in about 150 countries since 2005-2006 and includes a couple of questions that can be used to assess food security and hunger. Based on the GWP results, Headey concludes that the number of food-insecure people worldwide during the 2007-2008 period decreased by 60-90 million.

While these results seem to stand in stark contrast to the results of the simulation studies, the difference mainly stems from the fact that the studies try to answer a couple of different questions; while most simulation studies try to assess the change in poverty or food insecurity that might have occurred if food prices increased, everything else being equal (*ceteris paribus)* (Tiwari and Zaman 2010 is an exception), Headey (2011) estimates the actual change in food insecurity. Thus, Headey finds that the positive effect of economic growth in low- and middle income countries on food security outweighed the negative effect of food price increases during the 2007-2008 period.

However, when Headey uses the GWP survey results to estimate the change in food security that would have occurred from changes in food prices alone, *ceteris paribus*, results are very similar to those of the simulation studies, with the number of food-insecure individuals rising by 123 million. Put differently, both the survey based and simulation studies indicate that had not world food prices increased as they did in 2007-2008, another hundred million or so people would have been lifted out of food insecurity.

The latter results also point to the limitations of measuring the welfare effects of higher food prices solely by estimating the number of people crossing a given threshold in terms of income or calorie intake. Even if economic growth in developing regions would continue to lift people out of poverty and hunger, increases in food prices may still have detrimental welfare effects by lowering real incomes (as is the case with the general equilibrium studies reporting changes in GDP discussed above). Simulation studies also find that higher food prices tend to increase the poverty gap (i.e., pushing the already poor deeper into poverty), as well as moderate poverty (i.e., using a higher poverty line of $2.5/day) (Ivanic and Martin 2008; Wodon *et al*. 2008; de Hoyos and Medvedev 2011). On the other hand, most of the welfare impact studies reviewed above estimate direct, short-term effects, disregarding the secondary or indirect effects of higher food prices, primarily increases in the wage rate for rural unskilled labor and increases in agricultural output (yields) due to higher prices. The limited empirical evidence there is on the elasticity of agricultural wages with respect to prices suggests that wage responses are relatively small, especially in the short run (Ravallion 1990; Boyce and Ravallion 1991). Still, studies that do include the dynamic effects on wages find that these effects can substantially decrease the poverty impacts of higher food prices (Ivanic and Martin 2008; de Hoyos and Medvedev 2011).

Finally, although the aggregate response to the 2007-2008 food price spike was a pronounced increase in agricultural output, this supply response was largely concentrated to high and middle income countries (Christiaensen 2009). This finding is consistent with the historical evidence of low supply elasticities in countries in Sub-Saharan Africa (e.g., Thiele 2003).Moreover, the households currently most vulnerable to food price increases are likely to be less responsive to prices due to, e.g., a lack of access to credit needed to increase inputs, a less developed infrastructure for marketing, etc (Jayne *et al*. 2010). The possibility of mitigating the negative impacts from future increasees in biofuel demand on the world's poor is, therefore, highly dependent on our ability to instigate agricultural growth among smaller landholders in low-income countries through public sector investments (e.g., in infrastructure and agricultural research and development) and a stable and supportive policy environment (Jayne *et al*. 2010).

3.3.3 The global rush for land and effects on land rights and rural livelihoods

Since the beginning of this century, we have been witnessing what may be characterized as a global rush for farmland, with deals for the outright purchase, lease, or concessions of land in developing countries totalling over 200 million hectares (Mha) worldwide, or close to five times the area of Sweden (Anseeuw *et al*. 2012).[9] The bulk of these deals were made in the wake of the 2007-2008 food price crisis – between October 2008 and August 2009 alone, close to 50 Mha of large-scale land acquisition deals were struck (Deininger *et al*. 2011) – and over half of this area was located in Sub-Saharan Africa (Anseeuw *et al*. 2012).

The increased demand for land to produce biofuel feedstock has contributed to this phenomenon both directly and indirectly: directly, as the production of biofuel feedstocks accounts for the largest share of land acquisitions, 40 percent of the area for deals where the purpose of the land use is known, see Figure 14; indirectly, as the underlying driver of the land rush has been an expectation that a tightening global market for agricultural commodities – driven by increasing populations, incomes, and biofuels demand – will drive up future returns from arable land.

[9] Due to the lack of transparency the exact scale of this phenomenon is difficult to gauge. This number, which refers to deals reported in media or research reports and complied by the Land Matrix project up until November 2011, is likely to be an underestimate.

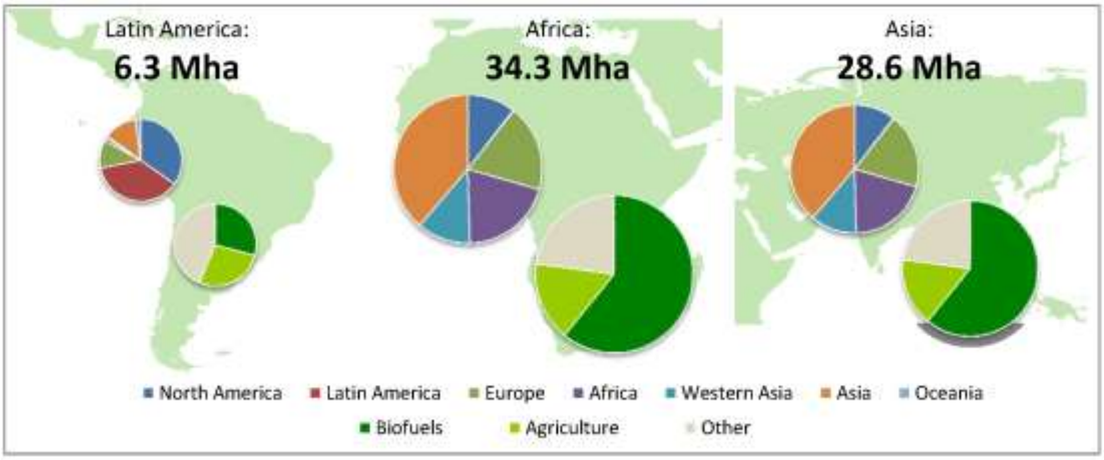

Figure 14. Distribution of land acquisitions between different regions where the investments originate from and between different planned uses of the land (others include forestry, industry, mining, tourism and miscellaneous). The data presented pertain to a subset of all reported large-scale land deals, not solely those relying on media reports but those cross-referenced from different sources. Adopted from Anseeuw et al.

While much media attention has been focusing on large-scale acquisitions by land (or water), in scarce countries like South Korea or Saudi Arabia or in emerging economies like China and India, the reality is more nuanced; private entities (companies and investment funds) account for the major share of land deals and national elites (politicians, civil servants, local business people) making investments targeted at domestic, rather than export, markets play important roles (Anseeuw *et al*. 2012; Cotula 2012).

While increased interests and investments in developing country agricultural potentials might have positive impacts on local livelihoods, there is overwhelming evidence that the results of the global land rush to date have been largely negative, leading to a widespread loss of access to land and other vital resources (e.g., water and housing) for local communities, with insufficient or non-existent compensation, and with women being disproportionally hard hit (Deininger *et al*. 2011; German *et al*. 2011; Anseeuw *et al*. 2012).

There are a number of reasons for this development. First, acquisitions often claim to target 'marginal' or 'unused' land but in reality, not much land fits into that description; for obvious economic reasons, most acquirers have prioritized land that is highly suitable for agriculture (i.e., fertile, well-watered or with good rainfall) with access to infrastructure and consumer markets (Anseeuw *et al*. 2012). Even in the cases where such land is not already under cultivation, it is likely to be collectively owned and used by local communities for grazing, hunting, harvesting forest products or shifting cultivation. Such lands often constitute the major asset of rural communities and their appropriation may have seriously adverse impacts on livelihoods, especially for the poorest households, pastoralists and forest dependent communities (Deininger *et al*. 2011; Anseeuw *et al*. 2012).

Second, many planned investments were not technically viable or investors lacked sufficient expertise, leading to many projects failing or falling far behind schedule. As a consequence, "local people had often suffered asset losses while receiving few or none of the potential benefits" (Deininger *et al*. 2011, p. xxxiii). In other cases, e.g., in Nepal and Uruguay, acquisitions have been purely speculative and have solely served to fuel land price inflation (Anseeuw *et al*. 2012).

Finally, developing country governments have been incapable or unwilling to harness the potentially positive force of investments to further strategic development plans and have instead offered acquirers land for little or no rent in an *ad hoc* manner, largely bowing to investor interests (Deininger *et al*. 2011; Anseeuw *et al*. 2012). Underlying these failings are existing power structures and the lack of functioning institutions in many host countries, legislative frameworks at both national and international levels that favor investor interests and large-scale commercial agriculture enterprises, leading to a neglect of the land rights of the rural poor and sidelining the involvement of smaller agricultural holders (Deininger *et al*. 2011; Anseeuw *et al*. 2012).

3.3.4 Summarizing the effects on food prices, food security, poverty and land rights

We have reviewed the evidence on the extent to which an increased demand for land for biofuels impact world food prices and in turn poverty, hunger, and the right for rural populations worldwide to their lands. We conclude that there is ample evidence of an increased integration between energy and agricultural markets due to the increased use of biofuels and the channel through which this integration primarily functions is through an increased competition for arable land.

That the price increases of the 2007-2008 period had significant effects on the welfare of low- and middle-income countries around the world is also clear, as the majority of these countries – and the majority of households in these nations – are net sellers of food. Again, methods and results vary between studies, but most estimates suggest that around 100 million people that would otherwise have been lifted out of poverty and hunger due to economic growth remained poor and undernourished because of these food price increases.

While there are a couple of studies that try to assess the subsequent impact on future poverty and hunger rates (Fischer *et al*. 2009; Cororaton *et al*. 2010), the results of these studies must be viewed as highly speculative (given the methodological challenges of even estimating the effect of recent price increases on these indicators). These effects are likely to be influenced by factors other than biofuel demand, such as poverty reduction through general economic growth and the success of improving yields and market access for small holders in low-income situations.

Still, given the current state of knowledge, it seems clear that most developing countries will stand to lose due to the higher demand for biofuels even if improved economic conditions may mean that this loss is not manifested through extreme poverty or malnutrition. Further, as long as there is scant progress on issues such as the strengthening of the resource rights of rural people (e.g., through legal recognition of land rights, including common lands), the empowerment of small producers (e.g., through contract farming arrangements with land investors) is likely to have negative impacts on the development (Jayne *et al*. 2010; Deininger *et al*. 2011; Anseeuw *et al*. 2012), making land use decision processes more transparent, inclusive and accountable since large-scale acquisitions of land in developing countries are driven by an increased demand for land for bioenergy.

A couple of final points are worthy of inclusion. The above discussion of the welfare impacts of biofuel use has been focusing solely on the increased levels of food prices. However, changes in price volatility can also have large welfare implications, especially for the poorest of the poor (FAO *et al*. 2011). Babcock (2011) suggests that the way in which current biofuel support policies have been implemented in the U.S. (mainly through the use of mandates) have been contributing to increased price volatility, whereas correctly formulated biofuel policies and the increased integration between energy and agricultural markets would actually have helped decrease the volatility of agricultural prices.

Second, most scenario studies show relatively modest impacts on world food prices from future increases in global biofuel demand and consequently, given the continued strong economic growth in developing regions, associated welfare impacts can most likely be expected to be limited. However, the size of biofuel demand shocks that these model exercises implement, are still minor compared to the bioenergy demand that might result from the combination of rising oil prices and stringent climate policies. Most studies model demand increases of around one to a few EJ/year[10], with concomitant land demand for biofuels in the order of 10-20 Mha, whereas long-term climate stabilization scenarios put global bioenergy demand in the order of 100-300 EJ/year, demanding some 500-1 000 Mha of land (Berndes *et al*. 2003). From the current studies, it is difficult to draw any clear conclusions on the impacts that such a large demand for land for biofuels might have on global agricultural markets and the welfare of the world's poor.

[10] Havlík *et al* (2011) and Msangi *et al* (2007) are exceptions, with a total biofuel demand shock of 11.6 EJ and 6.2 EJ, respectively, but the latter study also estimates much larger impacts on food prices (see Table 8).

Table 6. Summary of a selection of studies estimating the effects of recent and future demand for biofuels on agricultural commodity prices.

	Increase in the price of…						In year…	In…	Model	Size of		Share…		Price multiplier (%/EJ)	
Reference	Wheat	Corn	Soybean	Rice	Oilseed	Sugar			type	shock (EJ)	ethanol	biodiesel	2nd gen.	Corn	Oilseeds
Historic:															
		7.0%[a]			9.6%	5.5%	2006	EU	CGE	0.9	67%	33%	0%	0.12	0.33
(Birur *et al*. 2008)		8.7%[a]			5.9%	5.1%	2006	USA	CGE	0.9	67%	33%	0%	0.15	0.21
(Babcock 2011)	0.6%	2.6%	0.9%	-0.1%			2006	USA	PE	0.0	100%	0%	0%	-0.96	
	3.9%	15.2%	1.3%	0.1%			2007	USA	PE	0.1	100%	0%	0%	1.58	
	5.0%	11.7%	4.6%	0.9%			2008	USA	PE	0.3	100%	0%	0%	0.37	
	6.1%	23.0%	2.8%	0.7%			2009	USA	PE	0.5	100%	0%	0%	0.48	
	10.1%	26.8%	5.0%	1.2%			2010	USA	PE	0.6	100%	0%	0%	0.43	
(Rajagopal *et al*. 2007)		21.0%					2006	USA	PE	0.4	100%	0%	0%	0.48	
(Hausman *et al*. 2012)		8.6%					2007	USA	SVAR	0.1	100%	0%	0%	0.59	
(CBO 2009)		15-21%	16-27%				2008	USA	PE	0.8	100%	0%	0%	0.22	
(Birur *et al*. 2008)		5.9%[a]			6.4%	11.0%	2006	Brazil	CGE	0.9	67%	33%	0%	0.10	0.22
(Rajagopal *et al*. 2009)		5-13%	2-7%				2006	World	MPE	0.6	95%	5%	0%	0.16	
		15-28%	10-20%				2007	World	MPE	0.9	93%	7%	0%	0.26	
(Rosegrant 2008)[b]	18%	20%		9%			2007	World	PE	1.6	80%	20%	0%	0.16	
(Roberts and Schlenker 2009)	37.5%						2009	World	PE	1.1	100%	0%	0%	0.35	
Scenarios:															
(Britz and Hertel 2011)[c]	2%			1%	48%	1%	2015	EU	CGE	0.9[d]	0%	100%	0%		0.56
(Kretschmer *et al*. 2009)[c]	6.5%	4.0%			4.3%	5.1%	2020	EU	CGE	1.4[d]	60%	40%	0%	0.05	0.07
	6.2%	4.2%			7.3%	5.6%	2020	EU	CGE	1.4[d]	65%	35%	0%	0.04	0.14
(Blanco-Fonseca *et al*. 2010)	8.3%	22.2%	0.5%		10.5%	20.7%	2020	EU	PE	0.5	74%	26%	0%	0.62	0.82
	12.5%	8.0%			19.5%		2021	EU	PE	1.2	28%	72%	17%	0.23	0.22
(Chen *et al*. 2011)	7.4%	24.1%	19.7%	5.0%			2022	USA	MPE	3.2	100%	0%	58%	0.08	
	3.1%	-5.6%	0.3%	-1.0%			2022	USA	MPE	3.2	100%	0%	98%	-0.02	
(Hayes *et al*. 2009)	9.9%	22.3%	11.0%	4.0%			2022	USA	PE	2.2	94%	6%	64%	0.11	
	20.5%	46.9%	21.3%	9.7%			2022	USA	PE	3.0	96%	4%	47%	0.16	
(OECD 2008)	5.3%	7.5%[a]			3.1%	-2.0%	2015	World	PE	0.8	46%	54%	0%	0.19	0.07
	8%	13%[a]			7%	0%	2015	World	PE	1.8	46%	54%	29%	0.15	0.07
(Blanco-Fonseca *et al*. 2010)[c]	2.1%	1.5%			2.3%	-2.1%	2020	World	PE	1.0	27%	73%	20%	0.06	0.03

Table 6 (continued). Summary of a selection of studies estimating the effects of recent and future demand for biofuels on agricultural commodity prices.

	Increase in the price of…						In year…	In…	Model	Size of		Share…		Price multiplier (%/EJ)	
Reference	Wheat	Corn	Soybean	Rice	Oilseed	Sugar			type	shock (EJ)	ethanol	biodiesel	2nd gen.	Corn	Oilseeds
(Chakravorty *et al*. 2011)			14.2%[e]				2010	World	PE	-	-	-	-		
			31.1%[e]				2025	World	PE	0.8	67%	33%	15%		
			1.3%[e]				2010	World	PE	0.3	45%	55%	21%		
			5.4%[e]				2025	World	PE	0.8	67%	33%	48%		
(Kretschmer *et al*. 2009)	1.6%	1.6%			0.3%	1.3%	2020	World	CGE	1.4[d]	60%	40%	0%	0.02	0.01
	2.1%	2.0%			1.8%	1.9%	2020	World	CGE	1.4[d]	65%	35%	0%	0.02	0.04
(Msangi *et al*. 2007)[c]	30.0%	41.5%			76.0%	66.5%	2020	World	PE	6.2	76%	24%[f]	0%	0.09	0.52
	21.0%	29.5%			45.0%	49.5%	2020	World	PE	6.2	76%	24%[f]	-	0.06	0.31
	16.0%	24.0%			43.0%	43.0%	2020	World	PE	6.2	76%	24%[f]	-	0.05	0.29
(Rosegrant *et al*. 2008)	8.0%	26.0%			18.0%	12.0%	2020	World	PE	1.7	82%	18%	0%	0.19	0.62
	20.0%	72.0%			44.0%	27.0%	2020	World	PE	3.7	83%	17%	0%	0.23	0.71
(Timilsina *et al*. 2010)	1.1%	1.1%		0.8%	1.5%	9.2%	2020	World	CGE				0%		
		3.3%					2030	World	PE	11.6	-	-	0%		
		4.9%					2030	World	PE	11.6	-	-	40%		
		5.7%					2030	World	PE	11.6	-	-	60%		
		2.5%					2030	World	PE	11.6	-	-	0%		
		0.0%					2030	World	PE	11.6	-	-	40%		
		1.7%					2030	World	PE	11.6	-	-	60%		
(Prins *et al*. 2011)		14.0%			10.0%	0.7%	2030	World	CGE	0.7[d]	-	-	0%		
(Schmidhuber 2007)	0.40%	0.40%		0.50%		9.80%	2030	World	-	0.2	100%	0%	0%	0.02	
	0.60%	2.80%		1.00%		1.10%	2030	World	-	0.1	100%	0%	0%	0.32	
	0.90%	3.40%		1.20%		11.30%	2030	World	-	0.3	100%	0%	0%	0.12	

[a] Results for 'coarse grains'.

[b] Based on reported share of increase in world prices in 2007 due to expansion of biofuel demand between 2000-2007 and data on real commodity price changes between 2000-2007 from (World Bank 2012).

[c] Approximate price increases taken from the following graphs: Figure 2 (Britz and Hertel 2011), Figure 8 (Kretschmer *et al*. 2009), Figure 6 (Banse *et al*. 2008), Figure 3.2 (Blanco-Fonseca *et al*. 2010), and Figure 6 (Msangi *et al*. 2007).

[d] Projected biofuels demand as mandated by the Renewable Energy Directive, taken from (EC 2007).

[e] Weighted average of cereal and meat prices.

[f] Assumes that all expansion of biofuels demand in the EU is for biodiesel.

3.4 POLICY DEVELOPMENT RELATED TO LAND-USE CHANGE

As a result of a rapidly growing bioenergy market, a large number of national and international policies[11], standards[12] and certifications[13] have been developing worldwide. Most initiatives originate from the European Union or the Unites States where biofuel policies were early developed as the renewable industry expanded. Certifications and policies have the opportunity of positively influencing environmental and social aspects of biofuel production. There is a common understanding that policies and certifications have been playing and will continue to play an important role to streamline and harmonize international biofuel development in a way that minimizes environmental impact if proper traceability and monitoring components are included in the schemes. A large number of biofuel certifications exist today that target different social and environmental aspects of the production process which are developed by different organizations. Land-use change aspects of biofuel production are often issued through certifications, but may be more or less detailed. It should also be noted that land-use change in particular is regulated by other more legally binding laws, such as national governmental regulations on how land should be used, not by independent certifications which are usually voluntary.

The initiatives presented in this chapter are a selection of the various schemes in use today. All certifications presented are those that to some extent deal with land-use changes.

3.4.1 The Renewable Energy Directive (RED)

The Renewable Energy Directive (RED) was adopted by the European Commission in early 2009 and has since then been a basis for policy development, research and activities linked to the implementation of the Directive. The RED sets binding national targets for renewable energy use (which correspond to the overall EU target at a 20% share of energy from renewable sources), as well as a target of 10% renewable fuels for transport by 2020 for each member state. Included in the directive is a set of sustainability criteria with which biofuels and bioliquids have to comply in order to count towards the national targets. It is also indicated that the Commission will return to the question of the need for a sustainability scheme for energy uses of biomass, other than biofuels and bioliquids. The sustainability criteria include a target for greenhouse gas emission savings from the use of biofuels and bioliquids. The methodology for calculating the greenhouse gas emissions from the production and use of biofuels and bioliquids include emissions from carbon stock changes caused by land-use change. Land-use change is also touched upon in several ways throughout this Directive. Thus, land-use changes in the form of carbon stock changes e.g., when a forested area is cut down and converted to energy crop production are to be accounted for in the RED, as well as in the greenhouse gas calculations of biofuel production.

The conventional way of comparing biofuel chains with fossil fuels is to perform a greenhouse gas-related approach as pointed out in the RED. Even if biofuel use affects other aspects than the GHG

[11] A policy can be described as a rule of thumb to guide decisions and achieve a certain outcome. One example is the Renewable Energy Directive.
[12] A standard is a process of establishing common norms for a certain process.
[13] Certification means the conformation of certain characteristics of product or an organization. Certifications are often reviewed by an external independent organization.

balance, the carbon-based GHG approach is the universal approach in use today. In order for biofuels to become superior to fossil fuel based transport fuels from a GHG perspective, the carbon offset generated through the displacement of fossil fuels with biofuels must exceed the carbon storage and carbon sequestration that are halted due to direct or indirect land use (van Stappen, 2011).

In 2010, the EC launched a report on indirect land-use changes linked to biofuels. Since indirect land-use changes are by their nature highly uncertain, the scientific estimates are also highly diverse. Since the scientific results vary to such an extent, the EC has suggested four possible options on how to deal with land-use changes related to GHG emissions, namely (EC, 2010)

- take no action for the time being, while continuing to monitor,
- increase the minimum greenhouse gas saving threshold for biofuels,
- introduce additional sustainability requirements on certain categories of biofuels,
- attribute a quantity of greenhouse gas emissions to biofuels reflecting the estimated indirect land-use impact.

Introducing a specific iLUC factor for dedicated fuels, point four in the above list, is quite challenging as this also has to handle so-called “leakage effects[14]”. In October 2012, the European Commission put forward a proposal to introduce iLUC-factors for certain biofuel crops.

According to the RED, undesirable land-use changes can be categorized into land-use change from bioenergy crop production from the following: i) highly biodiverse land and ii) land with high carbon stock. Since the carbon stock of different land types depends on various factors, the RED aims at avoiding emissions from LUC by a general exclusion of some land types and by means of a minimum emission reduction target. Concerning the exclusion of certain land types, it is widely agreed that some land types such as forested lands, peatlands and wetland are particularly carbon rich and, therefore, do not automatically qualify for bioenergy production (Lange 2011). Concerning the minimum GHG reduction, biofuels must reduce emissions compared with a fossil fuel comparator of at least 35 % to qualify for the emission reduction target. This means, for example, that biofuels produced from land with high carbon content before a land-use change is taking place are less likely to reach the 35 % threshold.

Figure 15 illustrates the importance of the indirect land-use change emissions in European biofuel policy.

[14] Carbon leakage occurs when there is an increase in GHG emissions in one region/country as a result of GHG savings in another region/country.

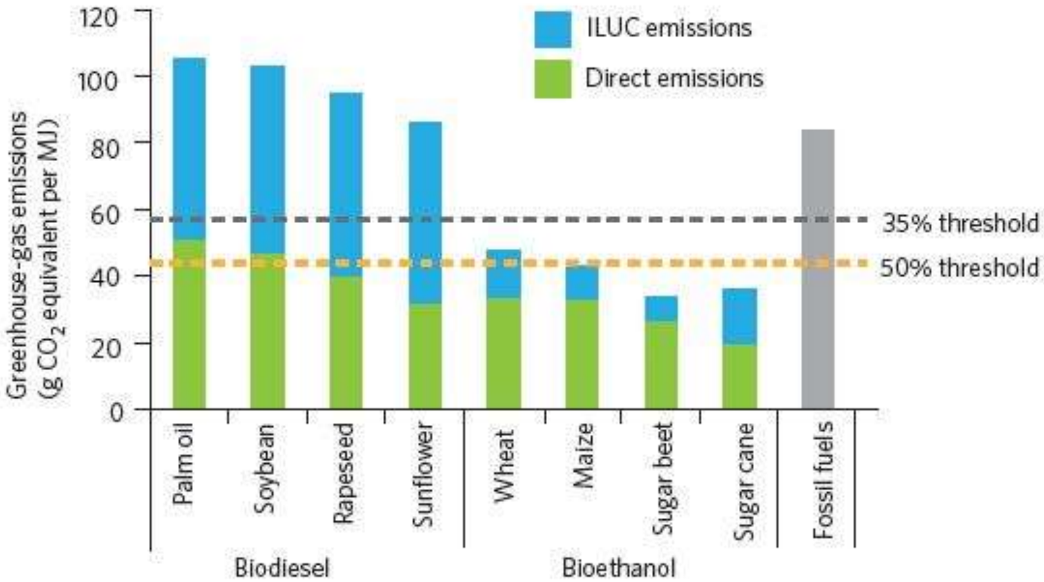

Figure 15. Greenhouse gas emissions from direct and indirect land-use changes for biodiesel and bioethanol from different feedstock. The blue bars illustrate additional emissions when iLUC is considered, making the thresholds specified in the Renewable Energy Directive heavily dependent on the indirect land-use change emissions. (Renssen 2011).

During 2012, the practical implementation in Sweden of the RED includes developing national greenhouse gas calculation tools for the emission savings criteria according to the Directive. The implementation means that organizations which produce or consume liquid or gaseous biofuels must use the method specified in the RED to calculate the greenhouse gas reduction of their sustainable biofuel quantities. The term "sustainable quantities" means biofuel quantities which fulfill the sustainability criteria issued by the Swedish Energy Agency, originating from the RED GHG methodology. Organizations which cannot prove that their biofuel qualify for the reduction target (i.e. biofuels that do not reach the emission reduction target of 35 %) are not eligible for the tax exemption for the particular biofuel. In order to arrive at a sustainability decision, the economic operator must have a system that ensures sustainability in the entire production chain and that is audited by an external auditor. The economic operator must also apply for a sustainability decision at the Swedish Energy Agency.

3.4.2 Certification initiatives – biofuel crops

Regulation for assessment of conformity for fuel ethanol (INMETRO)

The objective of the Brazilian INMETRO regulation is to promote exports and avoid technical barriers to trade. The initiative will develop a program for the certification of biofuels, initially only applicable for ethanol used for transport. The scope of the regulation is crop production and biofuel processing (Huertas *et al*, 2010).

Roundtable of sustainable Palm Oil

The objective of the Roundtable of Sustainable Palm Oil (RSPO) is to promote the growth and use of sustainable palm oil and develop standards for palm oil. The criteria included in the certification, which has been applied since 2008, cover major economic, social and environmental aspects, including direct land-use change issues. Indirect land-use change is excluded from the scheme. Principal countries using the scheme include Indonesia, Malaysia and Colombia where most of the world's palm oil is produced. It is a broad certification scheme but is solely focused on the production of palm oil and does not cover the transport or processing of the products. Several national interpretations of the scheme have been developed in such countries as Ghana and Thailand that address key concerns at the national level, similar to the national interpretations of the Renewable Energy Directive in Europe. If needed, the national laws can be complemented by the higher certification benchmark.

RSPO plans to include greenhouse gas emission calculations in its scheme and propose a way to handle greenhouse gas emissions related to land-use change (Scarlat and Dallemand, 2011).

Roundtable for Responsible Soy Production

The RTRS (Roundtable for Responsible Soy Production) is developed to promote responsible soy production, trade and processing. The standard includes five principles including environmental responsibility (Scarlat and Dallemand, 2011). However, land-use change issues were not included in the original standard but are included today since a voluntary RTRS certification scheme has been developed that complies with the RED (RTRS 2010).

Bonsucro (former Better Sugarcane Initiative)

Bonsucro has been developed in order to promote sustainable sugarcane production and reduce negative impacts on social, economic and environmental aspects. It includes a GHG calculation methodology which includes emissions associated with dLUC. Emissions related to iLUC are not included due to the lack of methodology and data. Bonsucro has submitted an application to the European Commission for its certification to be recognized as a voluntary scheme. If this becomes successful, importers of sugarcane ethanol to the EU will be able to show compliance with the RED (Scarlat and Dallemand, 2011).

3.4.3 Certification initiatives – European initiatives

Renewable Transport Fuels Obligation (United Kingdom)

The Renewable Transportation Fuels Obligation (RTFO) applies to road transportation fuels in the United Kingdom. The certification includes reporting on emissions from dLUC. Biofuel suppliers under the RTFO must also report on the land-use change history of the biofuel, if known, as expressed in the Renewable Energy Directive (Scarlat and Dallemand, 2011).

Sustainable production of biomass (The Netherlands)

In the Netherlands, the Cramer Commission has developed principles for the sustainability of biomass for energy (called NTA 8080). The principles cover both liquid and solid biofuels. Emissions related to direct land-use change are included, whereas indirect land-use change emissions are not (Scarlat and Dallemand, 2011).

International Sustainability and Carbon Certification (Germany)

The International Sustainability and Carbon Certification (ISCC) cover six principles linked to the sustainable production of bioenergy. The GHG calculation methodology also addresses emissions from direct land-use change in line with the methodology used in the RED (Scarlat and Dallemand, 2011).

CEN/TC 383 Standard for sustainably produced biomass for energy applications

The European Committee for Standardization (CEN) has established the Committee CEN/TC 383 for "Sustainably Produced Biomass for Energy Applications". The CEN/TC 383 may also develop requirements additional to those found in the RED which may be classified as a standard more than a certification. Six working groups have been established, one of which is dealing with indirect land-use change (but also indirect effects on socioeconomic aspects and food) (Scarlat and Dallemand, 2011).

3.4.4 Certification Initiatives – International initiatives

Roundtable on Sustainable biofuels

Originally a Swiss initiative, the Roundtable on Sustainable Biofuels (RSB) includes 12 principles for biofuel production. The intention is to develop a methodology to issue indirect impacts, such as iLUC and food security. The RSB is also discussing the possibility of including iLUC in the existing GHG methodology (Scarlat and Dallemand, 2011).

California Low Carbon Fuel Standard

Adopted in 2007, the Low Carbon Fuel Standard (LCFS) is intended to reduce GHG emissions for transportation fuels. The actual calculation methodology includes direct and indirect effects as well as emissions, including land-use change effects. The standard includes a model by which GHG emissions from land-use changes can be calculated by using an emission factor based on the type of land conversion and its carbon storage (Scarlat and Dallemand, 2011).

The Council on Sustainable Biomass Production

The Council on Sustainable Biomass Production (CSBP) was developed in the United States and has been formulating a wide range of sustainability criteria. The GHG calculation methodology developed considers land conversion and is aimed at including also indirect land-use change (Scarlat and Dallemand, 2011).

Global Bioenergy Partnership

The Global Bioenergy Partnership (GBEP) was developed in 2005 by the members of G8 in order to develop an international framework for bioenergy sustainability. The partnership is intended to build consensus, identify synergies and encourage integration. Criteria formulated under the GBEP include land-use change issues. Indirect land-use change will be further refined (Scarlat and Dallemand, 2011).

United States Renewable Fuels Standard

In 2005, the Energy Policy Act established a Renewable Fuels Standard (RFS) that required blending biofuels into transportation fuels. The Act includes targets for blending advanced biofuels, as well as conventional biofuels. The U.S. Environmental Protection Agency is in charge of tracking renewable fuel volumes from production to end-use. The renewable fuel standard was updated in 2010 and now requires a reduction in GHG life cycle emissions by 20 % for 1st generation biofuels, 50 % for advanced biofuels and 60 % for cellulosic biofuels. According to the regulation, the GHG emission methodology should include all life cycle emissions for all fuels, including emissions from iLUC.

ISO/PC 248 Sustainability Criteria for Biofuels

The International Standards Organisation (ISO) will develop biofuel standards related to bioenergy production. The ISO/TC 248 Project Committee will address sustainability issues related to social, economic and environmental aspects.

Forest Stewardship Council (FSC)

The international organization Forest Stewardship Council (FSC) aims to promote sustainable forest management. The scheme primarily addresses biodiversity issues and focuses mainly on the use of wood and timber rather than on biomass for energy (Höglund and Gustavsson, 2011). The FSC standardization documents are in constant development, as opposed to static.

Verified Sustainable Ethanol Initiative

The Swedish company SEKAB, involved in ethanol production and development, has developed the Verified Sustainable Ethanol Initiative. The initiative includes land-use change aspects; it prohibits the deforestation of rainforests to preserve biodiversity and the requirements of land-use change (Scarlat and Dallemand, 2011).

Programme for the endorsement of Forest Certification Schemes (PEFC)

The PEFC scheme promotes sustainable forest management. It addresses primarily biodiversity, ecosystem services and social aspects of forest management practices (Höglund and Gustavsson, 2011).

3.4.5 The Swedish Forestry Act

Swedish forest policy has a long tradition and the most recent review of the National Forestry Act was implemented in 1994. Compared to earlier versions, the contemporary Swedish Forestry Act advocates a balance between forest production and conservation values. Two overarching goals of equal value were established; one for production and another for safeguarding biodiversity. The intention is to make forest management sustainable. In 2004, following the intensified debate of human-induced climate change, the Forestry Act was reviewed further and now has a stronger emphasis on climate issues. Main stipulations of the current Swedish Forestry Act:

- mandatory reforestation after final felling
- a ban on the felling of young stands
- an obligation for forest owners to carry out preventive control of insect pests
- special management regimes for valuable hardwood and upland forests
- a general duty of care for forest objects or sites of natural, historical or heritage value (KSLA 2009).

The Forestry Act focuses on biodiversity protection; general conservation considerations in the law include the establishment of buffer-zones along watercourses, limitations of clear-felling areas and the retention of tree-filled areas within the clear-cuts. Land-use change is not directly mentioned in the Forestry Act but is dealt with indirectly through mandatory reforestation and a ban on fellings of young stands. The Act does not directly pinpoint climate issues or land-use changes in the context of biofuel production (Agency 2010).

Use of the forest and forest residues for biofuel purposes is not explicitly regulated by forestry legislation in Europe, with the exception of the Swedish Forestry Act requiring a notice to the authorities before clear-cuts and when logging residues are removed from the forest. The Swedish Forestry Act also gives advice concerning nutrient compensation and ash recycling (Stupak 2007a). Changes in land use, for example final felling operations or soil preparation prior to forest plantations is in other words regulated indirectly in forest legislation; however, not for the purpose of tackling land-use change effects.

The environmentally protective structure in the Forestry Act is in other words constructed from a landscape-biodiversity perspective for the purpose of creating distribution corridors, habitats with long continuity and appropriate amounts of dead and decaying wood.

In Sweden, primarily two major forest certification schemes are recognized; the Forest Stewardship Council (FSC) and the Program for the Endorsement of Forest Certification Schemes (PEFC). FSC is one of the most commonly adopted forest certification schemes to encourage sound forest management at different levels and contain social, ecological and economic aspects.

3.4.6 Socio-Economic criteria in policy

As shown in Table 7, most schemes include socioeconomic aspects in one way or another. Socio-economic issues are mainly expressed in international agreements and conventions, and many existing schemes express or mirror such international frameworks. Several schemes use requirements from international conventions, such as the ILO Convention, as a minimum baseline. Despite the inclusion of socio-economic aspects in many schemes, there is a great need to harmonize and develop a common methodology to measure and monitor such aspects (IFPRI 2011).

Table 7. Overview over existing certification schemes and factors included. "x" indicates that the factor is included in the scheme. For example, the RED includes three aspects related to LUC, but not iLUC (adapted from Scarlat and Dallemand, 2011).

Factor	Scheme	RED	RSPO	RTRS	BSI	RTFO	NTA 8080	ISCC	RSB	CSBP	GBEP	US-RFS	LCFS
LUC	***General***	x	x	(x)	(x)	x	x	x	x		x		
	GHG calculation	x	x		x	x	x	x	x			X	
	Monito-ring	x				x	x						
iLUC	***General***											X	
	GHG calculation											X	x
Bio-diver-sity		x			x	x	x	x	x	x	x		
Socio-economic aspects	***Economic develop-ment***		x	x	x		x	x	x		x		x
	Social aspects	x	x	x	x	x	x	x	x		x		x
	Labour conditions		x	x	x	x	x	x	x	x	x		x

3.4.7 Fossil fuels and Land-Use Change

The research on anthropogenic land-use change effects are more or less exclusively directed towards the production and use of biomass, in particular the production of biofuels. The coupling of bioenergy demand to LUC has been receiving a great deal of attention from researchers and policy makers. In the debate and research so far, the LUC effects of fossil fuels production and use are scarce. Gorissen *et al.* argue that the exploration of fossil fuel resources also induces land-use change, especially in the case of the exploration of unconventional fossil fuels, such as oil shale and tar sands, which involve ecosystem disturbances in the form of surface mining and horizontal well drilling. As conventional fossil fuel reserves decrease, unconventional fossil fuels (e.g. tar sands) become increasingly profitable to exploit, thereby increasing the risk of negative land-use changes also from fossil fuels. Regulating only the LUC effects associated with biofuels (not fossil fuels as well as other uses of biomass) can disturb the transition towards renewable fuels because the commercialization of renewable resources might be decelerated. Further, it can be argued that using fossil fuels as a reference energy system for comparisons with bioenergy necessitates a more comprehensive knowledge and assessment of the LUC performance of fossil fuel systems (Gorissen *et al* 2010).

3.4.8 Additional studies addressing policy and LUC

Since there is a scientific uncertainty regarding the scale and severity of indirect land-use change related to biofuels for transport, there is a challenge in developing suitable policies. If the policy governance of sustainable bioenergy use were not based on full information regarding adverse effects caused by land-use change (including indirect land-use change or iLUC), political and economic decision-making would rely on uncertain bases and the consequences might result in welfare losses to the entire society. Not taking the aspect into account or underestimating the impact might create welfare losses; in addition, erroneous accounting of the impact (i.e. overestimation) might also generate losses because of missed opportunities.

Gawel and Ludwig (2011) analyze the approaches developed for taking indirect land-use change into account. While instruments for certification schemes have been developed to tackle direct effects, the methodology of iLUC accounting is still in its infancy. However, three main approaches for governing iLUC-related bioenergy may be identified: the impact-related approach (e.g., the introduction of universal LUC requirements for all types of biomass or by protecting hot spots of biodiversity), the product assignment approach (e.g., the introduction of iLUC-factors) and the general government approach (e.g., introducing options for the effect of lowering the pressure on land use or by demanding continuous information on the impacts to legislators). The assessment of the discussed or even already implemented approaches shows that there is no predominant method that performs well, is practicable, available or finds general societal consensus. The conclusion by Gawel and Ludwig (2011) is that ”a more preferable option under present circumstances might be to diminish bioenergy targets and to choose bioenergy pathways with minor land use conflicts (e.g. residues) in order to lower the pressure on land-use change processes in bioenergy-producing regions worldwide”.

Di Lucia *et al*. (2012) analyze ways to deal with iLUC of biofuel policies by learning from policy fields where similar dilemmas have been confronted in the past. A risk-indifferent, risk-taking, preventive and precautionary approach are identified for the cases studied of carbon dioxide capture and storage (CCS), nuclear power and radioactive waste, genetically modified organisms (GMO) and biofuels for transport (past experience of environmental impacts). The results show that a preventive approach appears to be the most practical choice in terms of effectiveness and stakeholder acceptance. However, it also involves the risk of treating scientific uncertainty as certainty.

Havlík *et al*. (2011) who analyze the global land-use implications of first and second generation biofuel targets provide the following policy recommendation: “Due to the complexity of the system with both possible positive and negative effects and since no clear technology champion for all situations exists, policy action should focus directly on the positive and negative, environmental and social effects linked with biofuel production, rather than on biofuel production itself”.

A study by Huertas *et al*. (2010), based on interviews with different stakeholders, indicates that the Brazilian ethanol sector may adopt sustainability certification requirements rather than leaving markets where such requirements become established and turning to markets with less stringent demands. At the same time, results indicate that so far, the certification initiatives have not resulted in any profound changes in the Brazilian ethanol sector. It should also be noted that since the additional costs associated with certification may imply that small producers are unable to afford complying with certification (UNCTAD, 2008), certification may favor large corporations.

Di Lucia (2010) examines whether the sustainability certification system in the EU Directive 2009/28/EC (EC, 2009) can effectively ensure sustainable production of biofuels outside the EU by means of a case study of Mozambique. The focus is on evaluating the mode of EU external governance, i.e., the way the scope of the EU policy is expanded to include non-EU countries partly based on interviews. According to Di Lucia (2010), the EU attempts to impose its rules and values on sustainable biofuels by using its leverage through trade. The study finds that Mozambican industry is interested in participating in the EU market to promote its application to the EU system, while the lack of participation in the decision-making process does not advance the idea of a legitimate and appropriate policy in Mozambique. Stronger emphasis on a network- oriented approach based on the substantial involvement of foreign actors and on international policy legitimacy is suggested as a way to move ahead. It should be noted that only five countries were directly involved in the EC consultation process, three of them (Brazil, Malaysia and Indonesia) with well-established bio-fuel industries. However, the developing world, where a policy for sustainable biofuels can have substantial positive impacts, but where the biofuel industry is still in its infancy, does not have the same opportunity to influence the policy process.

3.4.9 Summarizing policy measures and Land-Use Change

Policies, certifications and standardizations are crucial in order to prevent negative land-use change effects. Many certification initiatives and standards address the land-use change issue in a variety of ways; these initiatives are usually expressed as GHG performance requirements. However, based on research so far, lignocellulosic biofuels may produce less negative effects compared to biofuels from agricultural crops.

The wide range of biofuel feedstock and conversion options used while producing biofuels means that biofuel certification faces large difficulties in formulating relevant criteria. Major concerns relate to the impacts that may occur in terms of biodiversity loss, impacts on water, water quality, food prices and land-use change, to name a few.

Even if biofuel certification were to address indirect land-use change, it is unlikely that all indirect effects can be avoided since they can be driven by other factors, as well as their effect on producers across different markets. From a policy development point of view, it might be doubtful to define principles and criteria that pinpoint only one particular end-use (biofuels), but no other uses (e.g. food). During the agricultural or forest management phase of the biofuel crop, the end-use of the biomass is not always known, which means that there are great uncertainties in presenting policies that are likely to end up in biomass for energy purposes alone.

Some existing schemes are intended to prevent iLUC. The Renewable Energy Directive promotes biofuels made from wastes and non-food cellulosic materials. The U.S. Renewable Fuels Standard (US-RFS) and the California Low Carbon Fuel Standard promote advanced biofuels (biofuels based on cellulosic feedstock) with lower iLUC impacts.

Still, there is great inconsistency and low harmonization among existing schemes dealing with LUC. There are signs that the trend is towards a more coherent and international framework; several schemes aim to comply with the Renewable Energy Directive. Nevertheless, the increased internationalization and globalization of biofuel standards are necessary to secure complex problems such as the indirect effects of biofuel production. Harmonization is needed to streamline definitions, approaches and methodoloies among existing schemes which means that new definitions are needed to avoid suboptimization; definitions of, for example, forest land, high conservation value areas, marginal areas, etc. As indirect effects can have global dimensions across countries as well as across markets, rigid and functioning schemes must also act in a harmonized manner across countries and markets.

Finally, the efforts of making biofuel production sustainable should also go hand in hand with making agriculture and forestry sustainable as well.

3.4.10 Knowledge gaps

There is a need for fresh knowledge concerning the impact of different policy mechanisms. Without such knowledge, it might be difficult to judge if a certain policy will live up to its intentions or whether it would create other undesireable side effects. Moreover, such fresh knowledge is needed on the ways in which different policies and certifications work together on a global market. Various policy options on land use effects in policy making should be evaluated in order to assess the actual impact of a policy. Land-use change aspects in policies are often vague and impossible to properly measure, and a better understanding of how such aspects might be developed and integrated into policy making would be valuable.

The effects of having a great variety of national and international initiatives working independently of each other is also questionable compared to having a few comprehensive international frameworks. Future research might be helpful in developing policy options that are measurable, effective and that avoid goals that are conflicting with other LUC aspects.

4 SYNTHESIS

4.1 INTRODUCTION

Humanity is transforming the terrestrial ecosystems at a global scale through the use of land for the production of food, feed, fiber, timber, and bioenergy. This current land use is in many ways problematic for the environment and is far from sustainable (Rockström *et al*. 2009; Foley *et al*. 2011). However, it is important to remember that all land-use change does not have to be detrimental to environmental sustainability, as there are opportunities to transition from current land use to land use that is better for the environment, for example by increasing soil carbon and biodiversity. Nevertheless, there is a considerable risk that land use change from increased production of biofuels will increase the pressure on the environment, and this risk must be carefully managed.

It is also important to consider that abstaining from using biofuel implies using fossil fuels. While most research regarding sustainability issues from LUC relate mostly to biofuel production, it is likely that the extraction of fossil fuel resources also induces LUC impacts. For example, fossil fuel exploitation can pose direct and indirect threats to biodiversity when reserves are located in fragile or biodiverse areas, as well as contribute to greenhouse gas emissions during the extraction and processing of unconventional fuel sources, such as oil sands and shale gas.

This report shows that issues related to land-use change are widespread and difficult to overview and can impact the local, regional or global environment. Synthesizing the information from the previous chapters is, therefore, valuable.

4.1.1 Framework of the synthesis

Land-use changes due to biofuel production can have impacts locally, regionally or even globally and affect a variety of socio-economic and environmental aspects. The effects can occur instantaneously or emerge slowly during a long period of time. Moreover, land-use changes are affected by policies and stakeholder decisions on local, regional and global levels. With the literature review as a base, the synthesis will discuss aspects of land-use change from a broader perspective, taking into consideration society, policies and future biofuel development.

Figure 16 shows how land-use change influences a variety of aspects at a local, regional, and global context. Wherever biomass is produced, i.e. on land in Sweden or elsewhere in the world, the production is affected by local and distant factors and has local and distant effects. The actual land use (inner circle) affects future production capacity on that land (green arrow). The production affects the local environment (biodiversity, water and soils) and local economy, and the local context in turn affects production (blue arrows). Production also affects the climate and world market, and is affected by them (orange arrow). Distant effects (orange arrows) can be both direct (i.e., greenhouse gas emissions from land use) and indirect (i.e., affects land use in other countries in addition to food prices).

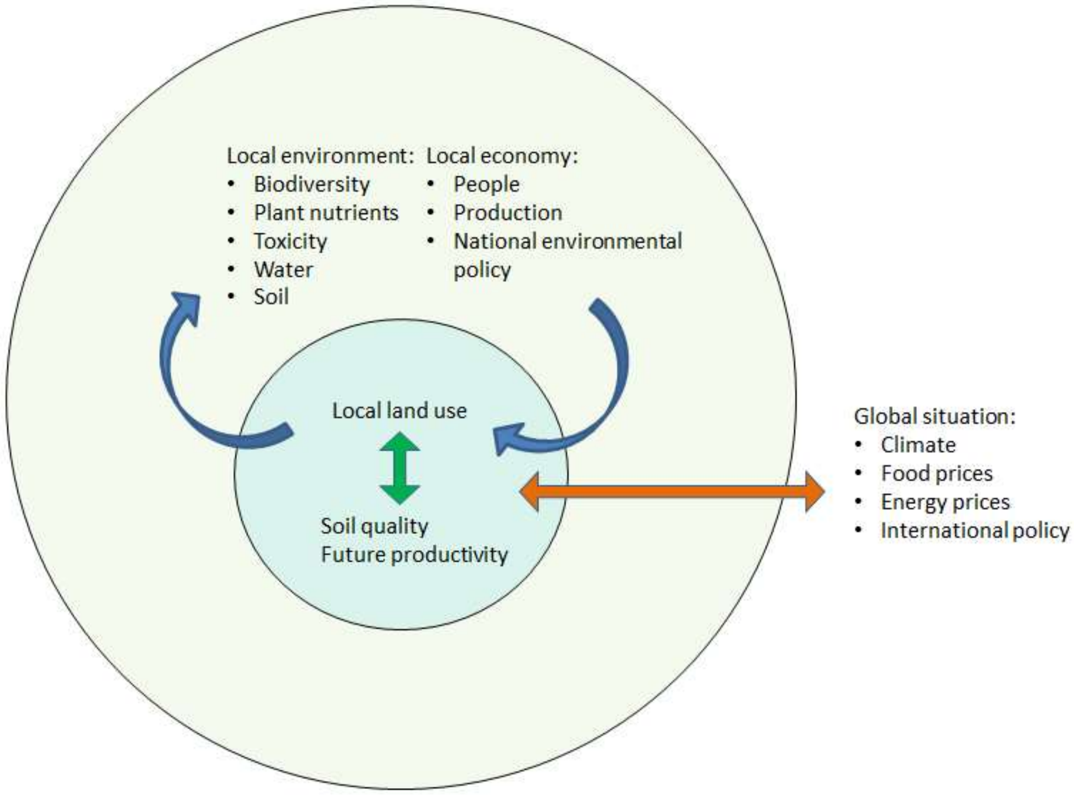

Figure 16. Local, regional and global land use implications.

4.2 MAJOR CHALLENGES IN THE CONTEXT OF LUC AND BIOFUEL PRODUCTION

In the literature review, a number of important land-use change issues associated with biofuel production at a local, regional and global scale, have been identified. These issues are either a result of direct or indirect land-use changes and have an impact on the local, regional and global scale.

4.2.1 Deforestation, forest management and climate change

Globally, deforestation is a major contributor to emissions of greenhouse gases. Through deforestation highly diverse ecosystems can be removed and replaced by agricultural lands. The carbon stock of standing biomass decreases and the crops that are grown after the land conversion often cannot reach the same carbon uptake rate. Carbon that has been stored in the soil can also be lost to the atmosphere. Other issues might arise as a consequence of the deforestation, such as losses of biodiversity, soil erosion, etc.

In Swedish forests, higher biomass production can be reached by increased harvests of tops, branches and stumps, as well as the nitrogen fertilization of forests and ash recycling. These changes in forest management affect greenhouse gas emissions from forest soils. Forest management practices for biofuel production would need to be monitored and modelled better so that management can be adapted to minimize negative environmental impacts. Moreover, changed forest practices may affect future forest growth, which is of major importance and requires long-term research.

A controversial issue related to the quantification of the climate effects of biomass is the timing of emissions. When biomass is burned, the carbon that is sequestered in a growing tree is released into the atmosphere over a short period of time. The carbon emitted by the combustion of biomass and loss of soil carbon must be "paid back" by being sequestered in new and growing biomass which – depending on the type of biomass – can take anything from two to 100 years to develop. Until now, this growth cycle has normally not been taken into account in greenhouse gas balances of biofuels.

In this report, we have also described the climate effects attributable to changes in albedo, aerosols and evapotranspiration; these changes may have both warming and cooling effects. For example, converting all year-round dark forest to agriculture that during the winter will be snow-covered produces a much whiter and more reflecting surface, which has a cooling effect. Cooling effects of a land-use change can be so large that it outweighs the warming effect of greenhouse gas emissions. However, since the estimations are based on rough approximations, while in reality local variations play a major role, the uncertainties are large.

4.2.2 Degradation of biodiversity

There is a general consensus among experts and governments worldwide that the biodiversity crisis, revealing the non-sustainable character of the human exploitation of natural resources, is one of the main threats to humanity (Millennium Ecosystem Assessment 2005, Rockström *et al*. 2009, Miljödepartementet 2009, UNEP 2011). Land-use changes caused by an expanded and intensified forestry or agriculture are nationally and globally among the most important threats to biodiversity. Intensified biofuel raw material production is qualitatively similar to other large-scale forestry or agriculture, but may have detrimental additive effects on biodiversity (for example, by using previously discarded biomass or land). The loss of species alters key processes of importance to the long-term productivity of ecosystems and is, therefore, an indicator of globally unsustainable land use. Environmentally motivated biofuel production must thus be established with great care or else, it may counteract its ultimate purpose of delivering sustainability.

As the literature review on biodiversity issues and land-use change clearly shows, the impact of biofuel raw material production on biodiversity may be negative, neutral or positive and the direction of the impact depends on the respective crop, habitat or land use replaced, landscape context of the plantations or group of organisms considered. Available studies give preliminary indications on the effects of different production systems on biodiversity, but further detailed studies are needed as a knowledge base for more solidly sustainable assessments of full-scale systems. The available literature is sparse in relation to the anticipated scale of bioenergy production (in Sweden as well as in other countries). It may not be surprising that research is lagging behind a quickly growing industry; however, it is urgent that every opportunity is taken to learn from new land use methods applied in experimental or full scale versions.

Further, there is emerging research about how biofuel feedstock can be produced in combination with the production of other ecosystem services on a win-win basis, but land use practice and policy still lag behind.

4.2.3 Nutrient leakage and removal

The market has become interested in using all parts of forest trees (stems, branches, tops, needles, and stumps). A change in the intensity of the production in a forest leading to the increased use of additional parts of the forest biomass has effects on growth potential, nutrient pools in forest biomass and soils, as well as the leaching of nutrients from forests.

The intensified use of forest biomass may also increase soil acidity and the loss of base cations leading to acidification. Studies have shown either no effect or a negative effect of intensive biomass harvesting on productivity in the short-term. Long-term studies are not yet available. Most studies show no unified effects on nitrate leaching after an intensified outtake of biomass (stems, branches and leaves/needles); however, studies reveal that the soil storage of nutrients is depleted, showing a negative nutrient budget which, in turn, will decrease long-term wood production capacity. The nutrient balance needs to be adjusted through one or several of the following recommendations: i) reconsidering the outtake of biomass, ii) returning ashes to the forest or iii) fertilizing the forest.

On Swedish agricultural land, an increased use of perennial energy crops instead of annual food crops may reduce direct environmental impacts from agriculture. The characteristics of perennial energy crops, such as low fertilizer input, low nutrient loss or high nutrient use efficiency values produces high efficiency in biomass production and less environmental effects from nutrient use. Perennial crops, such as reed canary grass and willow, have a year-round soil cover that is effective in reducing nutrient leakage. Perennial energy crops, if placed in suitable locations in the agricultural landscape, thus have the potential of contributing to reduced nutrient leakage from agriculture. However, the replacement of food production on Swedish agricultural land with perennial energy crops can cause indirect land-use change, moving the negative environmental effects of agricultural production to another location.

4.2.4 Challenges in quantifying iLUC

As the theory in iLUC-modelling is based on economic market reactions to an increasing demand for biofuels, as opposed to the more natural science-based quantification of direct changes, the quantification of climate change due to indirect land-use changes is very different from the quantification of direct changes.

According to current knowledge, there are great uncertainties as well as varying assumptions and methodologies among studies trying to quantify iLUC. The effects and methodological problems facing such studies are similar to research trying to measure biofuel contribution to rising food prices and poverty. What can be concluded, however, is that most studies acknowledge the existence of iLUC effects from biofuel production and that most studies present results that show that iLUC is relevant to the overall GHG balance. Based on existing knowledge, the European Commission published a proposal for greenhouse gas emissions from biofuels, but only for member state reporting purposes, in October 2012 to introduce so-called iLUC-factors into the EU calculation methodology. The proposal points out the direction for future biofuel production in Europe and that 2nd generation biofuels rather than agriculturally based biofuels should be promoted.

4.2.5 Contribution to rising food prices and poverty

Based on existing research, it seems clear that increased biofuel demand has had an impact on global food prices. Our literature review shows that a spectrum of different approaches to assess the contribution of biofuel use to the 2007-2008 food price crisis has been used. Although results vary, most studies find that a 30-50 percent increase in feedstock (corn and oilseeds) prices was due to rising biofuel demand. Note, however, that most studies focus on the effect of U.S. ethanol demand on domestic corn prices, while there is much less evidence of the impact of EU biodiesel demand on oilseeds and vegetable oil markets.

A number of studies have also tried to assess the consequences of further near-term increases of biofuels on agricultural markets. The results from these studies vary a great deal, partly depending on the methodologies used, with partial equilibrium models predicting larger price increases than general equilibrium models based on assumptions regarding the penetration of second generation biofuels which through their lower land use requirements assert less pressure on food prices.

It is also clear that higher food prices disproportionally impact the world's poor. Many of the world's poorest countries are net importers of food and will lose from higher food prices. Similarly, the major share of the population in these countries are net consumers of food and adapt to higher food prices by eating less, eating lower quality foods, or cutting back on other expenses. It is estimated that approximately 100-200 million people globally would have been lifted out of poverty and food insecurity had not agricultural commodity prices increased over the 2007-2008 period.

Increased global demand for land for biofuel production has also contributed – directly and indirectly – to large scale land acquisitions of farmland in developing countries by foreign investors, with the result that rural populations have lost access or rights to their land, leading to negative impacts on livelihoods and development opportunities.

4.2.6 Policy development

Today, numerous policies, certifications and standards exist to ensure the existence of sustainable biofuel production. Some policies were developed in an international context, such as the EU RED, while others target specific biofuels. Some policies are voluntary, while others are legally binding. Another controversy is that policies for sustainable biofuel production often presuppose the production of raw material for the specific biofuel end use. However, during plantation and growth, it is not always decided if a specific crop or tree will end up as biofuel, since it could also be harvested to produce such products as rape seed oil for human consumption or sawn wood products. Based on existing research, it is clear that a more harmonized approach is needed to consider the many effects of land-use change, regardless of the end use of the forestry and agriculture products.

4.2.7 Socio-economic aspects

Most human activities – agriculture, forestry, transport, housing and manufacturing – are all interconnected with the issue of land-use change. From a socioeconomic point of view, land-use change can be both positive and negative. Bioenergy production can create job opportunities in rural areas, while at the same time interfering with the traditional way of life, landscape preservation and recreational values. If increased biomass production in Sweden was to be successful, it would have to be developed in collaboration with all kinds of stakeholders, including farmers, rural communities, tourists, and hunters, in order to benefit from opportunities and manage conflicts.

4.3 VIEWS OF STAKEHOLDERS

As a part of this study, a workshop was arranged in order to discuss the results of the literature review in general and discuss stakeholder views on LUC issues in particular. In this section, stakeholder views are presented based on the project workshop.

4.3.1 How the biofuel industry perceives the LUC issue

Lack of political long-term decisions

In Sweden, as in other countries, the markets for biofuel have been backed up by short-term policies (up to four years) which have been creating markets for renewable fuels, such as the ethanol market and infrastructure. Because of the uncertainty regarding how greenhouse gas effective biofuels are compared to fossil fuels (and how to account for LUC-emissions), decision- makers are reluctant to make long-term commitments. The Swedish biofuel industry requests long- term government decisions on how to promote biofuels over the long run. An often stated argument as to why the introduction of second generation biofuels are not yet on the market at a larger scale is that investments are put to a halt because the lack of political will and direction. This is important considering LUC and iLUC-effects; the European biofuel industry has been waiting several years for a decision on how to account for iLUC (a proposal is now circulating for approval). Such uncertainty naturally creates concern and decision delays for the biofuel industry. This delay is especially problematic because even though there is scientific consensus that the environmental effects, including ILUC effects, are lower for 2nd generation biofuels than for currently produced fuels, the lack of political direction has delayed the development of 2nd generation biofuels.

The “food versus fuel debate” has implied that the benefits of using biofuel have been questioned from an ethical perspective. Also in this area, there is a need for clear policy decisions. As the biofuel industry expands, the growing market will have consequences for the global access to food, commodity markets and food prices, as we have seen during the literature review. On the other hand, an expanding biofuel industry may open up new opportunities for poor regions – small farms and rural communities – to benefit from the income from biofuel production and reduce unemployment.

A common argument for biofuel production in other countries is that it would promote economic and social development. However, such development must also be backed up by policies in order to prevent undesirable social and socioeconomic effects. Mechanisms to prevent such undesirable results include certifications and policy measures which need to be long-term in nature.

Policies that target a specific end use

Biofuel policies are usually developed with the view that a particular feedstock should be used for a particular end use and with the anticipation that this is already decided at the crop field which is not always the case in reality. Regardless of whether the crop ends up as biodiesel or cooking oil, the harvest of rape seed might create land-use change effects. The industry is critical of the fact that products are punished just because they are used for biofuels and advocates that all end uses should be subject to regulations regardless of their end use. Even if fossil fuel extraction that more or less depends on extraction techniques can induce land-use change, fossil fuels are rarely mentioned in the LUC-debate. With measures and regulations that target only biofuels, the industry feels it is penalized for actions it cannot reasonably influence.

Complex methodology accounting for LUC

Accounting for LUC and iLUC is a complex task. A common perception of the biofuel industry is that the methodology of accounting for LUC is complex and does not solve the problems. On the contrary, complex accounting might act as a barrier to biofuel expansion and create unnecessary administrative burdens. If the methodology becomes too complex, it would present obstacles to market development. Moreover, today's methodology to account for LUC is predominantly a matter of greenhouse gas calculations and does not take into account other effects described in this report, such as biodiversity, soil chemistry, albedo and so on. The methodology to account for LUC is also highly incertain, as described in the literature review.

Use of fertilizers

Since less land is needed to produce the same amount of feedstock, increased fertilization is one way of increasing biomass growth, thereby avoiding increased negative LUC impacts. In Sweden, there are a number of legal obstacles to the use of fertilizers in forests, coupled with risks associated with biodiversity and nutrient leakage. For the industry, costs can be excessive but industry nevertheless expects fertilization to be carried out if the legal framework were to become more cooperative.

4.3.2 Solutions

General long-term policies

The biofuel industry often suggests general and cost-effective measures to handle LUC-issues rather than specific targeted biofuel policies. Such policies should also be developed to take into consideration the agricultural and energy sectors at the same time. An important contemporary controversy is whether biofuel production involves both agricultural and energy polices or not, but the policies are not yet fully integrated on national or EU levels. In Sweden, there are agricultural subsidies available for the creation of buffer zones surrounding sensitive nature areas and catch crop cultivation to reduce nutrient leakage, but they are not allowed to be used for energy purposes.

Intensified cultivation

The intensified production of biomass can represent a solution to LUC issues related to climate impact. By producing more biomass on each hectare of land, for example by plantations of fast-growing trees, a higher demand for biofuels may not necessarily require expanding land use. Depending on soil quality and climatic region, yields can be increased in various ways (by new crop varieties, other management techniques or increased fertilizing). However, as there are risks concerning pesticides, nutrient leakage and biodiversity, such intensification can be controversial and requires careful management and policy development.

Efficient use of waste and more efficient post-harvest agricultural management

By using available resources such as food and waste more efficiently and by developing modern and effective management practices, LUC effects can be reduced. In developing countries, up to 25 % of the crops harvested can be destroyed due to poor processing and storage systems. By developing better storage conditions for food crops and bioenergy crops, less feedstock has to be discarded, which might decrease the pressure to expand agricultural areas. In industrialized countries, up to 25 % of the food ready for consumption is discarded. By reducing the amount of discarded food and by improving storage systems along the food logistics chain, less food must be produced, thereby reducing the need for using more arable land.

Energy crops on agricultural land.

On agricultural land, plantations of salix, poplar and energy grasses offer a number of opportunities:

- give considerably higher biomass yields than annual crops,
- contribute to reduced environmental impacts from agriculture through the reduced use of pesticides, less nutrient leakage and improved biodiversity in the agricultural landscape,
- increase carbon storage in vegetation and soils, and
- keep the potential to produce food on the land in the future.

Before Salix can become an important raw material for biofuel production, processing techniques must be further developed. Biofuel produced from salix and other cellulosic raw material is today regarded as 2nd generation fuels. Today, salix is mostly used in heat and power plants to produce heat and electricity.

However, all production of biofuels rather than food on agricultural land has an iLUC effect. As yields grow, the above-mentioned energy crops give less iLUC per MJ of biofuel than cereal and oil crops. Despite these opportunities, the production of cellulosic energy crops is not increasing, and Swedish salix is currently in decline.

Energy crops may have some slight positive effects on biodiversity compared to previous, large scale agriculture (by being perennial and structurally more diversified).

Moreover, there are indications that old traditional land management can be modernized to provide bioenergy feedstock sustainably, efficiently, and in harmony with ambitious biodiversity targets.

4.4 FUTURE OPPORTUNITIES AND MAJOR KNOWLEDGE GAPS

In the context of global land use challenges, the environmental and social risks in Sweden may seem trivial and even negligible. However, the situation in Sweden is the primary responsibility of Swedish actors, and there are many reasons for keeping high standards for Swedish biofuel production. The first conclusion for Sweden in a global context is that as we change land use, we should be careful not to export land-use-related environmental problems to other parts of the world. From a producer-perspective, this should primarily be done by increasing biomass production on Swedish land if such production can be carried out without harm to our environment (especially regarding climate change since this is an environmental effect with global consequences) and without reducing food production in Sweden.

Much of the negative effects on biodiversity and ecosystem services relate to the scale and intensity of the production system in question. This applies to Sweden as well as globally. Small scale production can create local benefits, such as energy security and employment opportunities. Policies should be developed that benefit small scale production and diversity in production systems.

4.4.1 Future research and knowledge gaps

A number of knowledge gaps have been identified throughout the literature review:

General knowledge gaps

- More research is needed on the impact of LUC related to fossil fuel production, e.g. when exploiting unconventional fossil fuels such as tar sands and shale gas so that the impact from fossil fuels can be compared to the impacts from biofuels (Gorissen *et al* 2010).
- More knowledge is needed in order to translate modern scientific knowledge to practice. Much is already known regarding greenhouse gas fluxes to and from the biofuel production system. The knowledge must be developed into guidelines that can be introduced into practice.
- Strategies for where to produce biofuels to avoid negative effects. More knowledge is needed concerning where biofuel production should be located, in what climatic region and on what types of soils.

Greenhouse gas emissions from soils

- Measurements of greenhouse gases. There is a large need for measuring carbon dioxide, methane and nitrous oxide from both forest and agricultural lands under various management practices. These measurements can yield a knowledge base for systems analyses in order to facilitate priorities between different biomass sources from a climate perspective.
- Studies on how forest growth is affected by harvest of tree tops, branches and stumps.
- Water transport of carbon from forests from a landscape perspective.
- How forests and biomass growth are affected by management using continuous tree cover compared to conventional clear-cutting.
- Knowledge of the potential for carbon storage in agricultural soils as land-use changes from annual food crops to perennial energy crops (including studies of roots and carbon turnover in soils).
- How nitrous oxide emissions are affected by different types of cropping systems and management activities.
- As organic soils are major emitters of greenhouse gases, it is important to find the best uses for these soils. There are indications that bioenergy production in some form can be a good management option for reducing GHG emissions from forested or agriculturally organic soils, but more research on this topic is needed.

Plant nutrient and other soil chemistry aspects

- Short-term studies on the effects of the outtake of biomass and the effects of wood ash recycling on forest production, as well as soil and water quality.
- Long-term studies of intensified biomass outtake from forests.
- Effects of the fertilization of forests.

- Continued work on how to transform experimental knowledge on wood ash recycling into practical guidelines in order to optimize the dose (i.e. optimizing fertilization doses yielding better biomass growth and at the same time avoiding negative environmental effects).
- Biochar is a promising option for long-term carbon sequestration in soils. Research on biochar is at an early stage and so far, there are more questions than answers regarding how biochar affects biomass productivity and long-term soil fertility.

Biodiversity

- There is a need for comprehensive studies on sustainability assessments of full-scale systems (given that biofuels production will be increased and more raw material is needed).
- Effects on biodiversity following the removal of stumps and clear-cutting residues. In more detail, studies are needed to reveal the effects on certain species after stump removal).
- Possibilities to grow willow beyond arable land need further investigation.
- Develop practices to adopt crop management to optimize biodiversity (finding optimal harvesting time and frequency, cutting height, species mix and so on).
- Particularly important is to initiate studies that lead to “win-win production systems”. Such research would give an indication of whether new species are worth introducing at a large scale.
- Analyses and assessments of landscape scale effects are needed. Today we have limited understanding of the proportion of land covered by energy crops vis-à-vis the total area and location of remaining habitat patches in relation to species area requirements and landscape fragmentation.
- Given future increased demand for biofuels, there is a need for studies of biodiversity effects taking into consideration the expected national increase in biomass use.

Socioeconomic aspects in Sweden

- More research is needed on how various stakeholders may be affected by land-use change caused by an increasing biofuel production.
- Ways to manage conflicts of interest between different stakeholders.
- How recreational values may be accounted for when deciding on biofuel strategies for the future.
- Research on how farmers can combine bioenergy production within their business.
- Socio-economic and technical barriers and opportunities for increased biomass production.

Biophysical and biogeochemical climate impacts of land-use change

- The influence of albedo effects of biomass production need to be further studied.
- The importance of compounds (aerosols) emitted from biomass burning (natural fires and human-induced fires) and how these compounds impact the climate.

- More research is needed to understand how the evapotranspiration balance in forests influences the climate.

- The feedback loops between climate and vegetation is important for the understanding of land-use change induced climate impacts. Feedback loops and their consequences are poorly understood.

Indirect land-use change

- Since estimates of iLUC are diverse, an increased knowledge on how to model iLUC is needed, using either economic or other types of models. This will improve our understanding of pros and cons, but never lead to a definite answer.

- Increased knowledge of historical cultivation systems and soil management on actual sites, as well as soil carbon, its past trend and other specific local conditions. Such understanding would reduce uncertainties in quantifying carbon stock.

- So far, only greenhouse gas emissions have been included in iLUC calculations. Including biophysical and biogeochemical impacts, such as changes in albedo, aerosols and evapotranspiration, could change the evaluation of iLUC climate impact effects.

Effects on food prices, food security, poverty and land rights

- The link between biofuel crop production and food prices is not known in detail. Biofuel production has had some effect on food prices, but the interdependence between food and biofuel crop cultivation is poorly understood.

- As is the case estimating iLUC-effects on climate, the effects of biofuel production on world poverty need to be further studied.

Policy development related to land-use change

- More research is needed on how biofuel policy mechanisms should be developed to consider several aspects of land-use change issues. Today, land-use change issues associated with greenhouse gases have been the subject of most studies.

- Measurable criteria should be further developed so that biodiversity and social aspects can be more easily measured and regulated through policy-making in the same way as greenhouse gas emissions (social life cycle analysis, biodiversity indicators, etc.).

- A major challenge is to develop policies that reach up to its intentions, while at the same time avoiding conflicts with other sectors and regulations.

The literature review and synthesis presented in this report have shown that land use on this planet is already placing high stress on ecosystems, atmosphere, soils and human life. Land-use change because of increased biofuel production is, therefore, at large risk of aggravating these problems. To avoid such risks and instead explore the opportunities for beneficial land-use change that exist, there is a need for continued responsible and sensitive collaboration between industry, policy-makers, researchers and local communities

REFERENCES

Abbott, P.C., C. Hurt, and W.E. Tyner. 2011. What's driving food prices in 2011? Farm Foundation. 47.

Abbott, P.C., C. Hurt, and W.E. Tyner. 2009. What's driving food prices? March 2009 update. Farm Foundation. 51.

Abbott, P. C., C. Hurt, and W. E. Tyner. 2008. What's driving food prices? Farm Foundation. 82

Adams, M.B., J.A. Burger, A.B Jenkins, and L. Zelazny, 2000. Impact of harvesting and atmospheric pollution on nutrient depletion of eastern US hardwood forests. For. Ecol. Manage. 138, 301-319.

Agency, S.F. 2010. "Skogsvårdslagstiftningen - gällande regler 1 september 2010."

Akselsson, C., J. de Jong, *et al*. 2011. Kan vi öka uttaget av skogsbränsle utan negativa konsekvenser för miljön? Uppsala, Sveriges Lantbruksuniversitet: 27.

Akselsson C. and O. Westling, 2005. Regionalized nitrogen budgets in forest soils for different deposition and forestry scenarios in Sweden. Global Ecology and Biogeography 14, 85-95.

Akselsson C., O. Westling, and G. Örlander, 2004. Regional mapping of nitrogen leaching from clearcuts in southern Sweden. Forest Ecology and Management 2002, 235-243.

Akselsson C., O. Westling, M. Alveteg, G. Thelin, A-M Fransson, and S. Hellsten, 2008. The influence of N load and harvest intensity on the risk of P limitation in Swedish forest soils. Science of the Total Environment 404, 284-289.

Alexandersson, H., U. Ekstam, and N. Forshed, 1986. Stränder vid fågelsjöar. Om fuktängar, mader och vassar i odlingslandskapet. Naturvårdsverket och LTs förlag, Stockholm.

Almers, R., 2009. Tillsats av biokol till en svensk sandjord, Kandidatarbete i biologi, Institutionen för mark och miljö, SLU Uppsala 2009, Examensarbeten, Available at: http://stud.epsilon.slu.se/667/1/almers_r_091202.pdf [29 May 2012]

Al-Riffai, P., B. Dimaranan, and D. Laborde, 2010. Global Trade and Environmental Impact Study of the EU Biofuels Mandate. Final Report March 2010, ATLASS Consortium. Study by IFPRI for the Directorate General for Trade of the European Commission.

Alvum-Toll, K., T. Karlsson, and H. Ström, 2011.Biochar as soil amendment– A comparison between plant materials for biochar production from three regions in Kenya, Degree project in Biology Agriculture Programme – Soil and Plant Sciences Institutionen för mark och miljö, SLU Uppsala 2011, available at: http://stud.epsilon.slu.se/2572/1/alvum_toll_k_etal_110509.pdf [29 May 2012].

Andersson P. 2000. Vad händer med skogsbrukets hänsyn efter avverkning? Undergraduate Thesis, Dept. of Conservation Biology, SLU, Uppsala.

Anderson-Teixeira K.J., S.C. Davis, M.D. Masters, and E.H. Delucia, 2009. Changes in soil organic carbon under biofuel crops. Global Change Biology Bioenergy 1(1):75-96.

Anseeuw, W., L. Alden Wily, L. Cotula, and M. Taylor. 2012. Land Rights and the Rush for Land: Findings of the Global Commercial Pressures on Land Research Project. International Land Coalition, Rome, Italy.

ARB, 2009. Final Regulation Order. Subchapter 10. Climate Change Article 4. Regulations to Achieve Greenhouse Gas Emission Reductions Subarticle 7. Low Carbon Fuel Standard. California Air Resources Board. Available at: www.arb.ca.gov/regact/2009/lcfs09/lcfscombofinal.pdf.

Aronsson K.A. and N.G.A. Ekelund, 2004. Biological effects of wood ash application to forest and aquatic ecosystems. Journal of Environmental Quality 33(5), 1595-1605.

Aronsson, P.G. and L.F. Bergström, 2001.Nitrate leaching from lysimeter-grown short-rotation willow coppice in relation to N-application, irrigation and soil type, Biomass and Bioenergy 21:155–164

Aronsson, P.G. Bergström L.F. and Elowson S. N. E., 2000. Long term influence of intensively cultured short rotation willow coppice on nitrogen concentration in grandwater. J. Environ Manage, 58:135-145

Arvidsson H., 2001. Wood ash application in spruce stands – Effects on ground vegetation, tree nutrient status and soil chemistry. Ph.D. Thesis, Swedish University of Agricultural Sciences. Silvestria 221, 1-33.

Augusto L., J. Ranger, Q. Ponette, and M. Rapp, 2000. Relationships between forest tree species, stand production and stand nutrient amount. Annals of Forest Science 57, 313-324.

Augusto L., J. Ranger, D. Binkley, and A. Rothe, 2002. Impact of several common tree species of European temperate forests on soil fertility. Annals of Forest Science 59, 233-253.

Babcock, B. A., 2011. The Impact of US Biofuel Policies on Agricultural Price Levels and Volatility. Issue Paper No. 35, International Centre for Trade and Sustainable Development (ICTSD). 38.

Bala G., K. Caldeira, A. Mirin, M. Wickett, C. Delire, and T. Phillips, 2006. Biogeophysical effects of CO_2 fertilization on global climate. Tellus B, Vol. 58, pp 620-627.

Bala G, Caldeira K, Wickett M, Phillips T, Lobell D, Delire C , Mirin A. 2007. Combined climate and carbon-cycle effects of large-scale deforestation. Proceedings of the National Academy of Sciences, Vol. 104, pp 6550.

Ballard, T.M., 2000. Impacts of forest management on northern forest soils. Forest Ecology and Management, 13: 37-42.

Banse, M., H. van Meijl, A. Tabeau, and G. Woltjer. 2008. Will EU Biofuel Policies Affect Global Agricultural Markets? European Review of Agricultural Economics 35(2):117-141.

Bardos P.R., B. Bone, Y. Andersson-Sköld, P. Suer, T. Track, and M. Wagelmans, 2011. Crop-damaged land. Remediation Journal, 21 (4) 11-33.

Batlle-Bayer L., N.H. Batjes, and P.S. Bindrapan, 2010. Changes in organic carbon stocks upon land use conversion in the Brazilian Cerrado: a review. Agriculture, Ecosystems and Environment 137, 47-58.

Bauen, A., C. Chudziak, K. Vad, and P. Watson, 2010. A causal descriptive approach to modelling the GHG emissions associated with the indirect land use impacts of biofuels. Final report. A study for the UK Department for Transport. E4tech.

Baum S., M. Weih, G. Busch, F. Kroiher, and A. Bolte, 2009. The impact of Short Rotation Coppice plantations on phytodiversity. Landbauforschung Volkenrode 59(3):163-170.

Beland-Lindahl, K. 2009. Skogens kontroverser. En studie om plats och politik i norra Sverige. Stockholm, Institutet för Framstidsstudier: 104.

Beland-Lindahl, K. and E. Westholm, 2010. "Food, Paper, Wood, or Energy? Global Trends and Future Swedish Forest Use." Forests 2(1): 51-65.

Berg, B. and G. Ekbohm, 1991. Litter mass-loss rates decomposition patterns in some needle leaf litter types. Long-term decomposition a Scots pine forest VII Can J Bot 69:1449-1456.

Bergh J., J. Witzell, L. Edenius, H. Grip, L. Gustafsson, T. Lind, A. Lindhegen, G. Mikusinski, R. Sathre, and J. Stendahl, 2010. Analys av miljö- och produktionsmål på region- och lanskapsnivå med storskalig intensivodling. Slutrapport 30399-1. Energimyndigheten, Eskilstuna.

Berg, Å. 1992. Factors affecting nest site choice and reproductive success of Curlews Numenius arquata on farmland. Ibis 134:44-51.

Berg, Å. and O. Kvarnbäck, 2005. Preferences for different arable field types among breeding farmland birds - a review. Ornis Svecica 15:31-42.

Berg, Å. & Gustafson, T. 2007. Meadow management and occurrence of corncrake Crex crex. Agriculture, Ecosystems & Environment 120:139-144.

Berglund, H. 2012. Biodiversitet. Chapter 10 in: de Jong J., Akselsson C., Berglund H., Egnell G., Gerhardt K., Lönnberg L., Olsson B. & von Stedingk H. 2012. Konsekvenser av ett ökat uttag av skogsbränsle. En syntes från Energimyndighetens bränsleprogram 2005 – 2009. Energimyndigheten, Eskilstuna. (In Swedish.)

Berndes, G., M. Hoogwijk, and R. van den Broek. 2003. The contribution of biomass in the future global energy supply: a review of 17 studies. Biomass and Bioenergy 25:1-28.

Betts, R.A., 2000. Offset of the potential carbon sink from boreal forestation by decreases in surface albedo. Nature 408, pp 187-190.

Bird, D., Cherubini, F., Cowie, A., Downing, M., Gustavsson, L., Kojakovic, A., Jungmeier, G., Möllersten, K., Pingoud, K., Rueter, S. 2009. IEA Bioenergy Task 38. Ten Years Of Analysing The Greenhouse Gas Balances Of Bioenergy Systems. 17th European Biomass conference and exhi-bition, Hamburg.

Birur, D. K., T. W. Hertel, and W. Tyner, 2008. Impact of Biofuel Production on World Agricultural Markets: A Computable General Equilibrium Analysis. GTAP Working Paper No. 53, Center for Global Trade Analysis, Department of Agricultural Economics, Purdue University.

Blanco-Fonseca, M., A. Burrell, H. Gay, M. Henseler, A. Kavallari, R. M'Barek, I. Pérez Domínguez, and A. Tononi. 2010. Impacts of the EU biofuel target on agricultural markets and land use: a comparative modelling assessment. Institute for Prospective Technological Studies, European Commission Joint Research Centre. JRC Scientific and Technical Reports, 130.

Boehmela, C. Lewandowskib, I. and Claupein, W., 2008. Comparing annual and perennial energy cropping systems with different management intensities, Agricultural Systems, 96 (1–3):224–236.

Boeker, E. and R. Van Grondelle, 1999. Environmental Physics. Second Edition. John Wiley & Sons.

Bonan, G., B. 2008. Forests and Climate Change. Forcings, Feedbacks and Climate Benefits of Forests. Science 320, 1444-1449.

Booth C., Eriksson H., Moller G., Pettersson F., Nykvist N., Olsson M., Lundin L., Persson H., Ahlstrom K. and Zundel P., 1990. Changes to forest management and silvicultural techniques necessitated by forest energy production. International Energy Agency/Bioenergy Agreement, Activity A1. Conventional Forestry Systems. Report 27, 1-153. Department of Forest Production, Swedish University of Agricultural Sciences, Uppsala.

Borgegård, Sven-Olov. Uppsala Universitet. Personal communication, 2012.

Boyce, J. K. and M. Ravallion. 1991. A DYNAMIC ECONOMETRIC MODEL OF AGRICULTURAL WAGE DETERMINATION IN BANGLADESH. Oxford Bulletin of Economics and Statistics 53(4):361-376.

Boyle, J.R., 1976. A system for evaluating potential impacts of whole-tree harvesting on site-quality. Tappi 59(7), 79-81.

Boyle, J.R., Phillips, J.J. and Ek, A.R., 1973. "Whole Tree" Harvesting: Nutrient Budget Evaluation. Journal of Forestry 71(12), 760-762.

BP, 2012. BP Energy Outlook 2030. London: 88. BP Statistical Review of World Energy. London: 48.

Bramryd, T. and Fransman, B.O., 1995. Silvicultural use of wood ashes - effects on the nutrient and heavy metal balance in a pine (Pinus sylvestris L.) forest soil. Water Air and Soil Pollution 85, 1039-1044.

Brinkman, H.-J., S. de Pee, I. Sanogo, L. Subran, and M. W. Bloem. 2010. High Food Prices and the Global Financial Crisis Have Reduced Access to Nutritious Food and Worsened Nutritional Status and Health. The Journal of Nutrition 140(1):153S-161S.

Britt, C., Bullard, M., Hickman, G., Johnson, P., King, J., Nicholson, F., Nixon P., and Smith N., 2002. Bioremediation - a review, bioenergy crops and bioremediation.

Britz, W. and T. W. Hertel. 2011. Impacts of EU biofuels directives on global markets and EU environmental quality: An integrated PE, global CGE analysis. Agriculture, Ecosystems and Environment 142(1-2):102-109.

Broome A., Clarke S., Peace A. & Parsons M. 2011. The effect of coppice management on moth ass-emblages in an English woodland. Biodiversity Conservation DOI 10.1007/s10531-010-9974-y.

Brown, M. E., F. Tondel, T. Essam, J. A. Thorne, B. F. Mann, K. Leonard, B. Stabler, and G. Eilerts. 2012. Country and regional staple food price indices for improved identification of food insecurity. Global Environmental Change (in press).

Brunner I., Zimmermann S. Zing A. and Blaser P., 2004. Wood-ash recycling affects forest soil and tree fine-root chemistry and reverses soil acidification. Plant and Soil 267, 61-71.

Bryngelsson, D. K., A. Åhlen, C. Azar, and U. M. Persson. 2012. The effect of food-price movements on African households. International Journal of Agricultural Resources, Governance and Ecology (in press).

Budikova, D., Hogan, C. 2010. Albedo. The Encyclopedia of Earth. www.eoearth.org/article/Albedo?topic=54300 Accessed 2013-01-22

Byrt,C. S., Grof, C. P.L. and Furbank R.T. , 2011. C4 Plants as Biofuel Feedstocks: Optimising Biomass Production and Feedstock Quality from a Lignocellulosic Perspective, Journal of Integrative Plant Biology 2011, 53 (2):120–135.

Cai, X., Zhang, X., Wang, D., 2011. Land Availability for Biofuel Production. Environmental Science & Technology 45, 334-339.

Campbell, J.E., Lobell, D.B., Genova, R.C., Field, C.B., 2008. The Global Potential of Bioenergy on Abandoned Agriculture Lands. Environmental Science & Technology 42, 5791-5794.

Carre, F., Hiederer, R., Blujeda, V., Koeble, R. 2009. Guide for the Calculation of Land Carbon Stocks on the Drawing on the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Document prepared as part of the Administrative Arrangement NO. TREN/D1/464-2009-SI2.539303, Ispra.

Carslaw, K., Boucher, O., Spracklen, D., Mann, G., Rae, J., Woodward, S., Kulmala, M. 2010. A review of natural aerosol interactions and feedbacks within the Earth system. Atmospheric Chemistry and Physics 10:1701-1737.

Caruso, A., Rudolphi, J. and Thor, G. 2008. Lichen species diversity and substrate amounts in young planted boreal forests: A comparison between slash and stumps of Picea abies. Biological Conservation 141:47-55.

Cederberg, B., Dahlberg, A., Egnell, G., Ehnström, B., Hallingbäck, T., Ingelög, T., Kjellin, P., Lennartsson, T., Lönnell, N., Schroeder, M., Thor, G., Thuresson, T., Tjernberg, M., Weslien, J. & Westling O. 2001. Skogsbränsle, hot eller möjlighet? - vägledning till miljövänligt skogsbränsleuttag. Skogsstyrelsens förlag, Jönköping. (In Swedish.)

Cederberg, C., Persson, M., Neovius, K., Molander, S., Clift, R. 2011. Including carbon emissions from deforestation in the carbon footprint of Brazilian beef. Environmental Science & Technology 45:1773-1779.

Cederberg, C., Flysjö, A., Ericsson, L. 2007. LCA av norrländsk mjölkproduktion (Life cycle assessment of milk production in northern Sweden). SIK-rapport 761, Institutet för livsmedel och bioteknik, Göteborg. ISBN 91-7290-237-X.

CBO. 2009. The Impact of Ethanol Use on Food Prices and Greenhouse-Gas Emissions. Pub. No. 3155, Congress of the United States, Congressional Budget Office (CBO), Washington, D.C., USA. 26.

Chakravorty, U., M.-H. Hubert, M. Moreaux, and L. Nøstbakken. 2011. Will Biofuel Mandates Raise Food Prices? Working Paper 2011-01, Department of Economics, University of Alberta. 50.

Chen, C.R. and Xu, Z.H., 2005. Soil carbon and nitrogen pools and microbial properties in a 6-year-old slash pine plantation of subtropical Australia: impacts of harvest residue management. Forest Ecology and Management 206, 237-247.

Chen, X., H. Huang, M. Khanna, and H. Önal. 2011. Meeting the Mandate for Biofuels: Implications for Land Use, Food and Fuel Prices. presented at "Agricultural Economics Conference", March 4-5, 2010 NBER WP-16697. National Bureau of Economic Research Working Paper Series. No. 16697: 60.

Cherubini, F., Peters, G.P., Berntsen, T., Strømman, A.H., Hertwich, E. 2011. CO2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. GCB Bioenergy, 3(5), 413-426.

Christiaensen, L. 2009. Revisiting the Global Food Architecture: Lessons from the 2008 Food Crisis. Discussion Paper No. 2009/04, United Nations University, World Institute for Development Economics Research (UNU-WIDER), Helsinki.16.

Chum, H., A. Faaij, *et al*. 2011. Bioenergy. IPCC Special report on renewable energy sources and climate change mitigation. O. Edenhofer, R. Pichs-Madruga, Y. Sokona*et al*. Cambridge, United Kingdom and New York, NY, USA., Cambridge University Press.

Ciaian, P. and d. A. Kancs. 2011. Interdependencies in the energy–bioenergy–food price systems: A cointegration analysis. Resource and Energy Economics 33(1):326-348.

Claussen, M., Brovkin, V., Ganopolski, A. 2001. Biogeophysical versus biogeochemical feedbacks of large‚Äêscale land cover change. Geophysical research letters, 28(6), 1011-1014.

Colbeck, I., Lazaridis, M. 2010. Aerosols and environmental pollution. Naturwissenschaften, Vol. 97, pp 117-131.

Cole, D.W. and Rapp, M., 1980. Elemental cycling in forest ecosystems. In: Reichle, D.E. (ed.) Dynamic properties of forest ecosystems. Cambridge University Press, London, pp. 341-409.

Conforti, P. 2004. Price transmission in selected agricultural markets. FAO commodity and trade policy research working paper No. 7, Food and Agricultural Organization of the United Nations (FAO), Rome. 91.

Cohen, W., & Levinthal, D. 1990. Absorptive capacity: A new perspective on learning and innovation. Administrative science quarterly, 35: 128-152.

Cornelissen, S., Dehue, B., Wonink, S., 2009. Summary of approaches to account for and monitor indirect impacts of biofuel production, PECPNL084225 / Ecofys

Cororaton, C. B., G. R. Timilsina, and S. Mevel. 2010. Impacts of Large Scale Expansion of Biofuels on Global Poverty and Income Distribution. Page 43 IATRC Public Trade Policy Research and Analysis Symposium, Stuttgart.

Costanza, R. and H. Daly. 1987. "Toward an Ecological Economics." Ecological Modelling 38: 1-7.

Cotula, L. 2012. The international political economy of the global land rush: A critical appraisal of trends, scale, geography and drivers. Journal of Peasant Studies 39(3-4):649-680.

Crutzen, P. J., Mosier, A. R., Smith, K. A. and Winiwarter, W., 2008. N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels, Atmos. Chem. Phys., 8:389–395.

Cullen, S.J., Montagne, C. and Ferguson, H., 1991. Timber harvest trafficking and soil compaction in western Montana. Soil Science Society American Journal 55, 1416-1421.

Dagernäs, D. 1996. Fåglar i odlingslandskapet – Biologisk mångfald och variation i odlingslandskapet. LRF, Sveriges Ornitologiska Förening and Swedish Board of Agriculture. (In Swedish.)

Dahlberg, A. 2011. Kontinuitetsskogar och hyggesfritt skogsbruk. Slutrapport för delprojekt naturvärden. Jönköping, Skogsstyrelsen: 96.

Dahlberg, A. & Stokland, J.N. 2004. Vedlevande arters krav på substrat – sammanställning och analys av 3600 arter. Skogsstyrelsen Rapport nr 7. (In Swedish.)

Dahlberg, A., Krikorev, M., Hansen, K., Jacobsson, S., Jeppson, M., Knutsson, T., Kuoljok, S., Larsson, K.-H., Nordén, B., Nitare, J., Svensson, S. & Tedebrand, J.-O. 2010. Fungi. In Gärdenfors U. (ed.). Rödlistade arter i Sverige 2010 – The 2010 Red List of Swedish Species. ArtDatabanken, SLU, Uppsala.

D´Almeida, C., Vörösmarty, C.J., Hurtt, G.C., Marengo, J.A., Dingman, S.L., Keim, B.D. 2007. The effects of deforestation on the hydrological cycle in Amazonia: a review on scale and resolution. International Journal of Climatology, 27, 633-647.

Daly, H. 1990. "Toward some operational principles of sustainable development." Ecological Economics 2: 1-6.

Dauber, J., Jones, M.B. & Stout, J.C. 2010. The impact of biomass crop cultivation on temperate biodiversity. Global Change Biology Bioenergy 2:289–309.

de Hoyos, R. E. and D. Medvedev. 2011. Poverty Effects of Higher Food Prices: A Global Perspective. Review of Development Economics 15(3):387-402.

Dehue, B., S. Cornelissen, *et al*. 2011. Indirect effects of biofuel production, Overview prepared for GBEP. Utrecht, Ecofys: 72.

Deininger, K., D. Byerlee, J. Lindsay, Andrew Norton, H. Selod, and M. Stickler. 2011. Rising global interest in farmland: Can it yield Sustainable and equitable benefits? , World Bank, Washington, D.C. 264.

de Jong, J., Akselsson, C., Berglund, H., Egnell, G., Gerhardt, K., Lönnberg, L., Olsson, B., von Stedingk, H. 2012. Konsekvenser av ett ökat uttag av skogsbränsle. En syntes från Energimyndighetens bränsleprogram 2007-2011. ER 2012:08. Energimyndigheten, Eskilstuna.

Dewbre, J., C. Giner, W. Thompson, and M. Von Lampe. 2008. High food commodity prices: will they stay? who will pay? Agricultural Economics 39:393-403.

DG Energy, 2010. The impact of land-use change on greenhouse gas emissions from biofuels and bioliquids. Literature review. An in-house review conducted for DG Energy as part of the European Commission's analytical work on indirect land-use change.

Di Lucia, L., 2010. External governance and the EU policy for sustainable biofuels, the case of Mozambique. Energy Policy 38 (11), 7395-7403.

Di Lucia, L., Ahlgren, S., Ericsson, K., 2012. The dilemma of indirect land-use changes in EU biofuel policy - An empirical study of policy-making in the context of scientific uncertainty. Environmental Science and Policy 16, 9-19.

Dimitriou, I., H. Rosenqvist, *et al*. 2011. "Slow expansion and low yields of willow short rotation coppice in Sweden; implications for future strategies." Biomass & Bioenergy 35(11): 4613-4618.

Dubois G.F., Vignon V., Delettre Y.R., Rantier Y., Vernon P. & Burel F. 2009. Factors affecting the occurrence of the endangered saproxylic beetle Osmoderma eremita (Scopoli, 1763) (Coleoptera: Cetoniidae) in an agricultural landscape. Landscape & Urban Planning 91(3):152-159.

Duncker P., Raulund-Rasmussen K., Gundersen P., de Jong J., Katzensteiner K., Ravn H.P., Smith M., Eckmüllner O., 2010. Report on time and scale integration of environmental impacts of forest management alternatives by use of generic modelling approaches: a case study of synergies and trade-offs between production and ecological services. Report EU project EFORWOOD.

Dynesius, M. 2012. Responses of bryophytes to wood-ash recycling are related to their phylogeny and pH ecology. Perspectives in plant ecology, evolution and systematics 14(1):21-31.

EC, 2012. Innovating for Sustainable Growth: A Bioeconomy for Europe. Brussels, European Commission: 9.

EC, 2011. Proposal for a Regulation of The European Parliament and of The Council on support for rural development by the European Agricultural Fund for Rural Development (EAFRD) 2011/0282 (COD). Brussels.

EC, 2010. "REPORT FROM THE COMMISSION on indirect land-use change related to biofuels and bioliquids." Brussels, 22.12.2010, COM(2010) 811 final.

EC, 2009. DIRECTIVE 2009/28/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Available at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=Oj:L:2009:140:0016:0062:en:PDF.

EC, 2007. Impact assessment of the Renewable Energy Roadmap - March 2007. Brussels, European Commission: 10.

Edenbrandt, A. 2012. Tillväxt, specialisering och diversifiering - hur har jordbruksföretagen förändrats de senaste 20 åren? AgriFood Economics Centre, Lund, 2012:2

Egnell, G., 2011. Is the productivity decline in Norway spruce following whole-tree harvesting in the final felling in boreal Sweden permanent or temporary? Forest Ecology and Management 261, 148-153.

Egnell, G. and Olsson, B. Växthusgaser. In: de Jong, J. (red.) 2012. Konsekvenser av ett ökat uttag av skogsbränsle. En syntes av Energimyndighetens forskningsprogram inom Skogsbränsle och Miljö 2005 – 2009. Energimyndigheten. Eskilstuna.

Egnell, G., Hyvonen, R., Hogbom, L., Johansson, T., Lundmark, T., Olsson, B.A., Ring, E., Öhlund J., von Sydow, F., 2007. Miljökonsekvenser av stubbskörd - en kunskapssammanställning och behovsanalys. Energimyndigheten. Rapport, 114 pp.

Egnell G., Bergh J., Dahlberg A., Rytter L. & Westling O. 2006. Miljöeffekter av skogsbränsleuttag och askåterföring i Sverige. En syntes av Energimyndighetens forskningsprogram 1997 till 2004. Rapport ER 2006: 44. Energimyndigheten, Eskilstuna. (In Swedish.)

Eisenbies, M.H., Burger, J.A., Aust, W.M. and Patterson, S.C., 2005 Soil physical disturbance and logging residue effects on changes in soil productivity in five-year-old pine plantations. Soil Science Society American Journal 69, 1833-1843.

Eliasson L., 2005. Effects of forwarder tyre pressure on rut formation and soil compaction. Silva Fennica 39, 549-557.

Elmer, W. H., and Pignatello, J. J. 2011. Effect of biochar amendments on mycorrhizal associations and Fusarium crown and root rot of asparagus in replant soils. Plant Dis. 95:960-966

Emanuelsson, U. 2009. Europeiska kulturlandskap, hur människan format Europas natur. FORMAS förlag, Stockholm.

Energimyndigheten, 2012. Hållbara biodrivmedel och flytande biobränslen under 2011. Eskilstuna, Statens Energimyndighet: 30.

Eriksson J. and Börjesson P., 1991 Wood ash in forests. FUD Report UB 1991/46. Vattenfall, Vällingby

Fabiosa, J. F., J. C. Beghin, D. Fengxia, A. Elobeid, S. Tokgoz, and Y. Tun-Hsiang. 2010. Land Allocation Effects of the Global Ethanol Surge: Predictions from the International FAPRI Model. Land Economics 86(4):687-706.

FAO, 2012. Low-Income Food-Deficit Countries (LIFDC) - List for 2012.Available at: www.fao.org/countryprofiles/lifdc/en/

FAO, 2008. Growing demand on agriculture and rising prices of commodities: An opportunity for smallholders in low-income, agricultural-based countries? Paper presented at the Thirty-first session of IFAD's Governing Council, 14 February 2008, Trade and Markets and Agricultural Development Economics Divisions, the Food and Agricultural Organization of the United Nations (FAO), Rome. 22.

FAO, IFAD, IMF, OECD, UNCTAD, WFP, t. W. Bank, t. WTO, IFPRI, and t. U. HLTF. 2011. Price Volatility in Food and Agricultural Markets: Policy Responses. World Bank, Washington, D.C. 68.

Fargione, J., Hill, J., Tilman, D., Polasky, S., Hawthorne, P., 2008. Land Clearing and the Biofuel Carbon Debt. Science 319, 1235-1238.

Ferm, A., Hokkanen, T., Moilanen, M. and Issakainen, J., 1992. Effects of wood bark ash on the growth and nutrition of a Scots pine afforestation in central Finland. Plant and Soil 147, 305-316.

Fischer, G., E. Hizsnyik, S. Preiler, M. Shah, and H. Van Velhuizen. 2009. Biofuels and Food Security. The Opec Fund for International Development (OFID) and International Institute for Applied Systems Analysis (IIASA). 228.

Florell, Erik, Swedish Biogas International AB. Personal Communication, 2012.

Foereid, B., Lehmann, J. and Major, J., 2011. Modeling black carbon degradation and movement in soil, Plant Soil 345:223–236.

Foley, J. A., N. Ramankutty, K. A. Brauman, E. S. Cassidy, J. S. Gerber, M. Johnston, N. D. Mueller, C. O/'Connell, D. K. Ray, P. C. West, C. Balzer, E. M. Bennett, S. R. Carpenter, J. Hill, C. Monfreda, S. Polasky, J. Rockstrom, J. Sheehan, S. Siebert, D. Tilman, and D. P. M. Zaks. 2011. Solutions for a cultivated planet. Nature 478:337-342.

Foley, J.A., Costa, M.H., Delire, C., Ramankutty, N., Snyder, P. 2003. Green surprise? How terrestrial ecosystems could affect earth's climate. Frontiers in Ecology and the Environment, 1(1), 38-44.

Forsum, Å. 2008. Nitrogen enrichment of a boreal forest, implications for understory vegetation. Doctoral Thesis, Department of Forest Genetics and Plant Physiology, SLU, Umeå.

Foster, B.L., Kindscher, K., Houseman, G.R. & Murphy, C.A. 2009. Effects of hay management and native species sowing on grassland community structure, biomass, and restoration. Ecological Applications 19(7):1884-1896.

Fransman, B. and Nihlgård, B., 1995. Water chemistry in forested catchments after topsoil treatment with liming agents in South Sweden. Water Air and Soil Pollution 85, 895-900.

Fredman, P., K. Sandell, *et al*. 2006. Friluftsliv i Förändring, Upplevelselandskap, planering och utveckling. Östersund, Friluftsliv i Förändring: 60.

Fritsche, U.R., Sims, R.E.H., Monti, A., 2010. Direct and indirect land-use competition issues for energy crops and their sustainable production - an overview. Biofuel Bioprod Bior 4, 692-704.

Froehlich, H.A., Miles, D.W.R. and Robbins, R.W., 1986. Soil bulk density recovery on compacted skid trails in central Idaho. Soil Science Society American Journal 49, 1015-1017.

Fuglestvedt, J.S., Berntsen, T.K., Godal, O., Sausen, R., Shine, K.P., Skodvin, T. 2003. Metrics of climate change: Assessing radiative forcing and emission indices. Climatic Change, 58(3), 267-331

Fuller, R.J. & Warren, M.S. 1993. Coppiced woodlands: their management for wildlife, 2nd ed. Joint Nature Conservation Committee, Petersborough.

Garcia, E. and Carignan, R., 2000. Mercury concentrations in northern pike (Esox lucius) from boreal lakes with logged, burned, or undisturbed catchments. Canadian Journal of Fisheries and Aquatic Sciences 57, (Suppl. 2) 129-135.

Gardiner, M.A., Tuell, J.K., Isaacs, R., Gibbs, J., Ascher, J.S. & Landis, D.A. 2010. Implications of Three Biofuel Crops for Beneficial Arthropods in Agricultural Landscapes. Bioenergy Research 3(1):6-19.

Gasparatos, A., P. Stromberg, Takeuchi, K., 2011. "Biofuels, ecosystem services and human wellbeing: Putting biofuels in the ecosystem services narrative." Agriculture, Ecosystems and Environment 142: 111-128.

Gawel, E., Ludwig, G., 2011. The ILUC dilemma: How to deal with indirect land-use changes when governing energy crops? Land Use Policy 28, 846-856.

German, L., G. Schoneveld, and E. Mwangi. 2011. Contemporary processes of large-scale land acquisition by investors. Case studies from sub-Saharan Africa. CIFOR Occasional Paper.

Giambelluca, T.W. 2002. Hydrology of altered forests. Hydrological Processes 16, 1665-1669

Glatzel, G., 1990. The nitrogen status of Austrian forest ecosystems as influenced by atmospheric deposition, biomass harvesting and lateral organomass exchange. Plant and Soil 128, 67-74.

Gohin, A. and F. Chantret. 2010. The long-run impact of energy prices on world agricultural markets: The role of macro-economic linkages. Energy Policy 38(1):333-339.

Gorissen, L., Buytaert, V., Cuypers, D., Dauwe, T., Pelkmans, L. 2010. "Why the debate about land-use change should not only focus on biofuels." Environ. Sci. Technol. 2010, 44(11): 4046–4049.

Goulding, K.W.T. and Stevens, P.A., 1988. Potassium reserves in a forested acid upland soil and the effects on them of clearfelling versus whole-tree harvesting. Soil Use Management 4, 45-51.

Grelle, A. 2011. Stubbrytningens effekter på skogsekosystemets koldioxidbalans. Slutrapport, projekt 30476-1. Energimyndigheten, Eskilstuna

Gundersen, P., Schmidt, I.K. and Raulund-Rasmussen, K., 2006. Leaching of nitrate from temperate forests - effects of air pollution and forest management. Environmental Reviews 14, 1-57.

Gustafsson, L. 2004. Skogsbränsleuttagets påverkan på naturhänsynen – en analys av nuläget. Slutrapport, Energimyndigheten. (In Swedish.)

Gustafsson, L., S. C. Baker, *et al*. 2012. "Retention Forestry to Maintain Multifunctional Forests: A World Perspective." BioScience 62(7): 633-645.

Gustafsson L., Dahlberg A., Green M., Henningsson S., Hägerhäll C., Larsson A., Lindelöw Å., Lindhagen A., Lundh G., Ode Å., Strengbom J., Ranius T., Sandström J., Svensson R. & Widenfalk O. 2009. Konsekvenser för kulturarv, friluftsliv, landskapsbild och biologisk mångfald. Faktaunderlag till MINT-utredningen. Report, SLU, Uppsala.

Gustavsson, R. & Ingelög, T. 1994. Det nya landskapet. Skogsstyrelsen, Jönköping.

Göransson, G. 1994. Bird fauna of cultivated energy shrub forests at different heights. Biomass and Bioenergy 6:49-52.

Hanson, C., J. Ranganathan, *et al*. 2012. The Corporate Ecosystem Service Review - Guidelines for Identifying Business Risks and Opportunities Arising from Ecosystem Change. Washington, World Resources Institute (WRI), World Business Council for Sustainable Development (wbcsd) and Maridian Institute: 48.

Hansson, H., & Ferguson, R. 2010. Factors influencing the strategic decision to further develop dairy production — A study of farmers in central Sweden. Livestock Science, 135(2-3): 110-123.

Hausman , C., M. Auffhammer, and P. Berck. 2012. Farm Acreage Shocks and Food Prices: An SVAR Approach to Understanding the Impacts of Biofuels. SSRN eLibrary: 29.

Hautala, H., Jalonen, J., Laaka-Lindberg, S. & Vanha-Majamaa, I. 2004. Impacts of retention felling on coarse woody debris (CDW) in mature boreal spruce forests in Finland. Biodiversity and Conservation 13:1541-1554.

Havlík, P., U. A. Schneider, E. Schmid, H. Böttcher, S. Fritz, R. Skalský, K. Aoki, S. D. Cara, G. Kindermann, F. Kraxner, S. Leduc, I. McCallum, A. Mosnier, T. Sauer, and M. Obersteiner. 2011. Global land-use implications of first and second generation biofuel targets. Energy Policy 39(10): 5690-5702.

Hayes, D. J., B. A. Babcock, J. F. Fabiosa, S. Tokgoz, A. E. Elobeid, T.-H. Yu, F. Dong, C. E. Hart, E. C. Chavez, S. Pan, and M. Carriquiry. 2009. Biofuels: Potential Production Capacity, Effects on Grain and Livestock Sectors, and Implications for Food Prices and Consumers. Journal of Agricultural and Applied Economics 41(02):465–491.

Headey, D. 2011. Was the Global Food Crisis Really a Crisis? Simulations versus Self-Reporting. IFPRI Discussion Paper 01087, International Food Policy Research Institute.

Headey, D. and S. Fan. 2008. Anatomy of a crisis: the causes and consequences of surging food prices. Agricultural Economics 39:375-391.

Headey, D., S. Malaiyandi, and S. Fan. 2010. Navigating the perfect storm: reflections on the food, energy, and financial crises. Agricultural Economics 41:217-228.

Hedin, J., Isacsson, G., Jonsell, M. & Komonen, A. 2008. Forest fuel piles as ecological traps for saproxylic beetles in oak. Scandinavian Journal of Forest Research 23: 348-357.

Helby, P., H. Rosenqvist, *et al*. 2006. "Retreat from Salix - Swedish experience with energy crops in the 1990s." Biomass & Bioenergy 30(5): 422-427.

Helldin, J-O 2008. Storskalig hamling av lövskog – en potentiell bioenergiresurs. Biodiverse 1: 8-9.

Helldin, J-O, Lennartsson, T & Emanuelsson, U. 2010. Biologisk mångfald och bioenergi i odlingslandskapet – en kunskapssammanställning. Report to Swedish Board of Agriculture. CBM/SLU, Uppsala. (In Swedish.)

Hellebrand, H.J., Strähle, M., Scholz, V. and Kern, J. 2010. Soil carbon, soil nitrate and soil emissions of nitrous oxide during cultivation of energy crops. Nutrient Cycling in Agroecosystems. 87:175-186.

Hellsten, S., Helmisaari, H-S., Melin Y., Skovsgaard, J.P., Wängberg, I., Kaakinen, S., Kukkola, M., Petersson, H. and Akselsson, C., 2009. Halter av baskatjoner, fosfor och kväve i stubbar i Sverige, Danmark och Finland. IVL Rapport B1855, Stockholm.

Hellsten, S., Akselsson, C., Hagberg, L. & Fuhrman, F., 2010. Effekter av skogsbränsleuttag på näringsämnesbalanser som underlag för avvägning mellan miljömål. Rapport B1880. IVL, Stockholm.

Helmisaari H-S., Holt-Hansen, K., Jacobson, S., Kukkola, M., Luiro, J., Saarsalmi, A., Tamminen, P. and Tveite, B., 2011. Logging residue removal after thinningin Nordic boreal forests: Long-term impact on tree growth. Forest Ecology and Management 261, 1919-1927.

Hendrickson, O.Q., Chatarpaul, L. and Burgess, D., 1989. Nutrient cycling following whole-tree and conventional harvest in northern mixed forest. Canadian Journal of Forest Research 19, 725-735.

Henningsson, M., S. Jönsson, *et al*. 2012. Vindkraftens påverkan på människors intressen. En syntesrapport. Vindval. Stockholm, Naturvårdsverket: 194.

Hertel, T.W. 2010. The Global Supply and Demand for Agricultural Land in 2050: A Perfect Storm in the Making? GTAP Working Paper No.63.

Hillier, J., Whittaker, C., Dailey, G., Aylott, M., Casella, E., Richter, G.M, *et al*. 2009. Greenhouse gas emissions from four bioenergy crops in England and Wales: Integrating spatial estimates of yield and soil carbon balance in life cycle analyses. Global Change Biology Bioenergy 1(4):267-81.

Hjältén, J., Stenbacka, F. & Andersson, J. 2010. Saproxylic beetle assemblages on low stumps, high stumps and logs: Implications for environmental effects of stump harvesting. Forest Ecology and Management 260:1149-1155.

Hoffmann, D. & Weih, M. 2005. Limitations and improvement of the potential utilisation of woody biomass for energy derived from short-rotation woody crops in Sweden and Germany. Biomass and Bioenergy 28:267-279.

Holmroos, S., 1993. Karaktärisering av vedaska. VU-E 93:E64. Vattenfall Utveckling AB, Ekraftteknik VI-EC.

Hultengren, S., Johansson, P., Croneborg, H. & Lönnell, N. 2006. Threatened biodiversity in wooded hay meadows on Gotland. Svensk Botanisk Tidskrift 100(3):176-194.

Huertas, D.A., Berndes, G., Holmén, M., Sparovek, G., 2010. Sustainability certification of bioethanol: how is it perceived by Brazilian stakeholders? Biofuels, Bioproducts & Biorefining 4 (4): 369-384.

Huikari O., 1951. Havaintoja ojitettujen rimpinevojen taimettumista ehkäisevistä tekijöistä. Suo: Mires and Peat 2, 1-4.

Huntington, T.G. and Ryan, D.F., 1990. Whole tree harvesting effects on soil nitrogen and carbon. Forest Ecology and Management 31, 193-204.

Hylander, L. D.,Günther, F. and Hansson, K., 2010. Climate saving soils with biochar.conference paper In: NJF seminar 430, Climate Change and Agricultural Production in the Baltic Sea Region, 4-6 May 2010 Sweden : Focus on Effects, Vulnerability and Adaptation.

Hyvönen, N.P., J.T. Huttunen, N.J. Shurpali, N.M. Tavi, M.E. Repo, P.J. Martikainen. 2009. Fluxes of nitrous oxide and methane on an abandoned peat extraction site: Effect of reed canary grass cultivation Bioresource Technology 100, 4723–4730.

Högbom, L., Ring, E and Jacobson, S., 2008. Långsiktiga effekter av asktillförsel resp. fördelning av avverkningsrester på hyggen mark- och markvattenkemiska effekter. Slutrapport 30636-1. Energimyndigheten, Eskilstuna.

Höglund J., Gustavsson, M., 2011. "Assessment of the RED sustainability criteria and their applicability for solid biofuels - A Swedish energy sector perspective." Elforsk report 11:68.

IEA, 2010. Bioenergy, Land-use change and Climate Change Mitigation. Lead authors: Göran Bern-des (Chalmers University of Technology) Contributing authors: Neil Bird (Joanneum Research) and Annette Cowie (National Centre for Rural Greenhouse Gas Research). IEA Bioenergy Ex-Co: 2010:03.IFPRI, 2011. "Assessing the Land-use change Consequences of European Biofuel Policies." David Laborde (IFPRI) ATLASS Consortium.

IPCC. 2007. Technical Summary. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Inventories, Vol 4, Japan.

ISO, 2010. Guidance on Social Responsibility. Geneva, International Organization for Standardization.

Isselstein, J., Jeangros, B. & Pavlu, V. 2005. Agronomic aspects of biodiversity targeted management of temperate grasslands in Europe – a review. Agronomy Research 3:139–151.

Ivanic, M. and W. Martin. 2008. Implications of higher global food prices for poverty in low-income countries. Agricultural Economics 39:405-416.

Jack, S., & Anderson, A. 2002. The effects of embeddedness on the entrepreneurial process. Journal of Business Venturing, 17: 467-487.

Jacobson, S. & Gustafsson, L. 2001. Effects on ground vegetation of the application of wood ash to a Swedish Scots pine stand. Basic and Applied Ecology 2:233-241.

Jacobson, S., Kukkola, M., Mälkönen, E.and Tveite, B., 2000 Impact of whole-tree harvesting and compensatory fertilization on growth of coniferous thinning stands. Forest Ecology and Management 120, 41-51.

Jayne, T. S., D. Mather, and E. Mghenyi. 2010. Principal Challenges Confronting Smallholder Agriculture in Sub-Saharan Africa. World Development 38(10):1384-1398.

Johansson, T., J. Hjältén, *et al*. 2009. Generell hänsyn och naturvärdesindikatorer - funktionella metoder för att bevara och bedöma biologisk mångfald i skogslandskapet. Solna, Världsnaturfonden WWF: 54.

Johnson, F. X., E. Pacini, *et al*. 2012. Transformations in EU biofuels markets under the Renewable Energy Directive and the implications for land use, trade and forests. Bogor, Indonesia, Center for International Forestry Research (CIFOR): 69.

Johnson D.W. and Curtis P.S., 2001. Effects of forest management on soil C and N storage: meta-analysis. Forest Ecology and Management 140, 227-238.

Johnson D.W. and Todd Jr. D.E., 1998. Harvesting effects on long-term changes in nutrient pools of mixed oak forest. Soil Science Society American Journal 62, 1725-1735.

Joki-Heiskala P., Johansson M., Holmberg M., Mattsson T., Forsius M., Kortelainen P. *et al*., 2003. Long-term base cation balances of forest mineral soils in Finland. Water, Air, and Soil Pollution 150, 255-273.

Jonsell M., Hansson J. & Wedmo L. 2007. Diversity of saproxylic beetle species in logging residues in Sweden – Comparisons between tree species and diameters. Biological Conservation 138: 89-99.

Jonsell, M. & Hedin, J. 2009. GROT-uttag och artmångfald – hur bör man ta hänsyn till vedskalbaggar? SLU Fakta Skog No 7/2009. (In Swedish.)

Jonsson, H. and J.-O. Thorstensson. 2012. Skogen vi ärvde - ”Sverige har redan en bra skogspolitik”. DN.se. Dagens Nyheters nätupplaga: 1.

Jonsson, L. and R. Uddstål. 2002. En beskrivning av den svenska skogsbärsbranchen. Vindlen, Vindelns försöksparker, Sveriges Lantbruksuniversitet: 25.

Junninen, K., Similä, M., Kouki, J. & Kotiranta, H. 2006. Assemblages of wood inhabiting fungi along the gradients of succession and naturalness in boreal pine dominated forests in Fennoscandia. Ecography 29:75–83.

Kahl, J.S., Fernandez, I.J., Rustad, L.E. and Peckenham, J., 1996. Threshold application rates of wood ash to an acidic forest soil. J. Envir. Qual. 25, 220-227.

Kanakidou, M., Seinfeld, J., Pandis, S., Barnes, I., Dentener, F., Facchini, M., Van Dingenen, R., Ervens, B., Nenes, A., Nielsen, C. 2005. Organic aerosol and global climate modelling: a review. Atmospheric Chemistry and Physics, 5(4), 1053-1123.

Kardell, L., 2010. Skogsenergiförsöken 1977-2008. Stubbrytningens mm effekter på markvegetation och skogsproduktion. Rapport 111. Institutionen för skoglig landskapsvård. SLU, Uppsala.

Karltun, E., Saarsalmi, A., Ingerslev, M., Mandre, M., Andersson, S., Gaitnieks, T., Ozolincius, R. and Varnagiryte-Kabasinskiene, I., 2008. Wood ash recycling – possibilities and risks. In: Röser, D.; Asikainen, A.; Raulund-Rasmussen, K.; Stupak, I. (Eds.). Sustainable Use of Forest Biomass for Energy. A Synthesis with Focus on the Baltic and Nordic Region (pp. 79–108). Springer.

Kasimir-Klemedtsson, Å. 2009. Lustgasavgång från jordbruksmark. Institutionen för Geovetenskaper, Göteborgs universitet.

Kellner, O. & Weibull, H. 1998. Effects of wood ash on bryophytes and lichens in a Swedish pine forest. Scandinavian Journal of Forest Research 13:76-85.

Keenan, R.J. and Kimmins, J.P., 1993. The ecological effects of clear-cutting. Environmental Reviews 1, 121-144.

Key, R.S. 1995. Invertebrate conservation and new woodland in Britain. In: Ferris-Kaan R. (ed.) The Ecology of Woodland Creation, Wiley, New York.

Khanna, M., Crago, C., 2011. Measuring Indirect Land-use change with Biofuels: Implications for Policy, Agricultural Policy Briefs. Dep of Agricultural and Consumer Economics. University of Illinois at Urbana-Champaign. March 18, 2011. APBR 11-01.

Kinell, G., T. Söderqvist, *et al*. 2009. Monetära schablonvärden för miljöförändringar. Stockholm, Naturvårdsverket: 132.

Kirby, K. 1993. Coppice restoration for nature conservation: how much and where? Pages 15-24 in: Proceedings Coppice Restoration Seminar in Cowes, Isle of Wight, 6th May 1993.

Kirschbaum, M., Whitehead, D., Dean, S., Beets, P., Shepherd, J., Ausseil, A.G.E. 2011. Implications of albedo changes following afforestation on the benefits of forests as carbon sinks. Biogeosci-ences, 8, 3687-3696.

Klemedtsson, L., von Arnold, K., Weslien, P., and Gundersen, P.: 2005. Soil CN ratio as a scalar parameter to predict nitrous oxide emissions, Global Change Biol., 11, 1142–1147,

Knoepp, J.D. and Swank, W.T., 1997. Forest management effects on surface soil carbon and nitrogen. Soil Science Society American Journal 61, 928-935.

Kofman, P., 1987. Wood ashes from chip fuelled heating plants: Chemical composition, possibilities of application. The Danish Institute of Forest Technology, Hørsholm, Denmark, p. 1-20.

Kretschmer, B., D. Narita, and S. Peterson. 2009. The economic effects of the EU biofuel target. Energy Economics 31, Supplement 2(0):S285-S294.

Kreutzer, K., 1995. Effects of forest liming on soil processes. Plant Soil 168-169: 447-470.

KSLA, 2009. "The Swedish forestry model." The Royal Swedish Academy of Agriculture and Forestry (KSLA), Stockholm, Sweden.

Kätterer, T., Bolinder, M.A., Andrén, O., Kirchmann, H., Menichetti ,L. 2011. Roots contribute more to refractory soil organic matter than above-ground crop residues, as revealed by a long-term field experiment. Agriculture, Ecosystems & Environment 141(1–2):184-92.

Lafond, G.P., Stumborg, M., Lemke, R., May, W.E., Holzapfel, C.B., Campbell ,C.A. 2009. Quantifying Straw Removal through Baling and Measuring the Long-Term Impact on Soil Quality and Wheat Production. Agronomy Journal 101(3):529-37.

Lal, R., 2008. Soil carbon stocks under present and future climate with specific reference to European ecoregions, Nutr Cycl Agroecosyst 81:113–127.

Lange, M. 2011. "The GHG balance of biofuels taking into account land-use change." Energy Policy 39 (2011) 2373 – 2385.

Larsson, S., Lundmark, T. & Ståhl, G. 2009. Möjligheter till intensivodling av skog. Slutrapport från regeringsuppdrag Jo 2008/1885.

Leemans, R., van Amstel, A., Battjes, C., Kreileman, E., Toet, S., 1996. The land cover and carbon cycle consequences of large-scale utilizations of biomass as an energy source. Global Environmental Change, 6: 335-357.

Ledare (2012). Skogen vi ärvde - Stärk skyddet av skogen vi ärvde. DN.se. Dagens Nyheters nätupplaga - huvudledare: 1.

Liebig, M.A., Gross, J.R., Kronberg, S.L., Phillips, R.L., Hanson J.D. 2010. Grazing Managements Contributions to Net Global Warming Potential: A Long-Term Evaluation in the Northern Great Plains. J. Environ.Qual. 39:799-809.

Likens, G.E., Bormann, F.H., Johnson, N.M., Fisher, D.W. and Pierce, R.S., 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed ecosystem. Ecological Monographs 40, 23-47.

Likens, G.E., Driscoll, C.T., Buso, D.C., Siccama, T.G., Johnson, C.E., Lovett, G.M., Fahey, T.J., Reiners, W.A., Rya,n D.F., Martin, C.W. and Bailey, S.W., 1998. The biogeochemistry of calcium at Hubbard Brook. Biogeochemistry 41, 89-173.

Lindelöw, Å. 2009. Insekter. Pages 89-97 in: Gustafsson, L., Dahlberg, A., Green, M., Henningsson, S., Hägerhäll, C., Larsson, A., Lindelöw, Å., Lindhagen, A., Lundh, G., Od,e Å., Strengbom, J., Ranius, T., Sandström, J., Svensson, R. & Widenfalk, O. 2009. Konsekvenser för kulturarv, friluftsliv, landskapsbild och biologisk mångfald. Faktaunderlag till MINT-utredningen. Report, SLU, Uppsala.

Lindroth A., Smith, B., Sykes, M., Wallander, H., Vestin P., 2012. Landsbygdsministern har fel – kalhyggen inte klimatsmarta. DN debatt. Dagens Nyheter. 2012-05-30.

Liska, A.J., Perrin, R.K., 2009. Indirect land use emissions in the life cycle of biofuels: regulations vs science. Biofuels, Bioproducts and Biorefining 3: 318-328.

Ljungberg, H. 2001. Jordlöpare som indikatorer vid övervakning av värdefulla naturmiljöer – Sammanställning och utvärdering av befintlig kunskap. (In Swedish.)

Lukkala, O.J., 1955. Soil improving substances and fertilizers as an aid to forest drainage. Metsätaloudellinen Aikakauslehti 8, 273-276.

Lukkala, O.J., 1951. Experiences from Jaakkoinsuo experimental drainage area. Commun. Inst. For. Fenn., 39: 1-53.

Lundmark, T., A. Nordin, *et al*. 2012. Skogen vi ärvde - ”Besluten om skogen måste beakta alla dess värden”. DN.se. Dagens Nyheters nätupplaga: 2.

Lövgren, S. and Westling, O., 2002. Modell för att beräkna kväveförluster från växande skog och hyggen i sydsverige. Rapport 2002:1. Institutionen för miljöanalys, SLU.

Mahli, Y., Roberts, J.T, Betts, R.A., Killeen, T.J., Li, W., Nobre, C.A. 2008. Climate Change, Deforestation and the Fate of the Amazon. Science 319: 169-172.

Makeschin, F., 1994. Efffects of energy forestry on soil. Biomass ans Bioenergy, 6: 63-79

Maljanen, M., B. D. Sigurdsson, J. Guðmundsson, H. Óskarsson, J. T. Huttunen, and P. J. Martikainen 2010. Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps. Biogeosciences, 7, 2711–2738, 2010

Mangoyana, R. B. and T. F. Smith. 2011. Decentralised bioenergy systems: A review of opportunities and threats. Energy Policy (39) 1286-1295.

Mann, L.K., Johnson, D.W., West, D.C., Cole, D.W., Hornbeck, J.W., Martin, C.W., Riekerk, H., Smith, C.T., Swank, L.M., Tritton, L.M. and Van Lear, D.H., 1988. Effects of whole-tree and stem-only clearcutting on postharvest hydrologic losses, nutrient capital, and regrowth. Forest Science 34: 412-428.

Manninen, K., R. Antikainen, *et al*. 2012. Indentification and quantification of indirect land and resource use changes – Challenges caused by expanding liquid biofuel production. Helsinki, Finnish Environment Institute: 72.

Marland, G, Schlamadinger, B. 1997. Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. Biomass and Bioenergy 13: 389-397.

Mattson, K.G. and Swank, W.T., 1989. Soil and detrital carbon dynamics following forest cutting in the southern Appalachians. Biol. Fertil. Soils 7, 247-253.

Mattsson, L., M. Boman, *et al*. 2008. "Så mycket är jakten värd." Svensk Jakt(5).

Mendham, D.S., O'Connell, A.M., Grove T.S. and Rance S.J., 2003. Residue management effects on soil carbon and nutrient contents and growth of second rotation eucalypts. Forest Ecology and Management 181, 357-372.

Merino, A. and Edeso, J.M., 1999. Soil fertility rehabilitation in young Pinus radiate D.Don. plantations from northern Spain after intensive site preparation. Forest Ecology and Management 116, 83-91.

Miljøministeriet, 2008. Skov- og Naturstyrelsen & Forsvarets Bygnings- og Etablissementstjeneste. Finderup Øvelsesplads Drifts- og Plejeplan 2008- 2022.

Millennium Ecosystem Assessment 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington DC Island Press. Available at. www.maweb.org/documents/document.354.aspx.pdf

Minot, N. 2011. Transmission of World Food Price Changes to Markets in Sub-Saharan Africa. IFPRI Discussion Paper 01059, International Food Policy Research Institute, Washington, D.C. 44.

Mitchell, D. 2008. A note on rising food prices. World Bank Policy Research Working Paper No. 4682 World Bank, Washington, D.C. 21.

Moe, B. & Botnen, A. 2000. Epiphytic vegetation on pollarded trunks of Fraxinus excelsior in four different habitats at Grinde, Leikanger, western Norway. Plant Ecology 151:143-159.

Moilanen, M., Silfverberg, K. & Hokkanen, T.J. 2002. Effects of wood-ash on the tree growth, vegetation and substrate quality of a drained mire: A case study. Forest Ecology and Management 171:321–338.

Mroz, G.D., Jurgensen, M.F. and Frederick, D.J., 1985. Soil nutrient changes following whole tree harvesting on three northern hardwood sites. Soil Science Society American Journal 49, 1552-1557.

Msangi, S., T. Sulser, M. W. Rosegrant, and R. Valmonte-Santos. 2007. Global scenarios for biofuels: Impacts and implications. Farm Policy Journal. Farm Policy Journal 4(2):1-9.

Munthe, J. & Hultberg, H., 2004. Mercury and methylmercury in runoff from a forested catchment – concentrations, fluxes and their responses to manipulation. Water, Air and Soil Pollution, Focus 4, 607–618.

Mälkönen, E., 1996. Tuhka kangasmetsien lannoitteena (In Finnish). Metsäntutkimuslaitoksen tiedonantoja 599, 21-26.

Møller, I.S., 2000. Calculation of biomass and nutrient removal for different harvesting intensities. New Zealand Journal of Forestry Science 30(1/2), 29-45.

Møller, H.B., Nielsen, L. & Christensen, T.B. 2007. Biogas production from different types of biomass and grass species from meadows. Pages 87-88 in: Nordic Association of Agricultural Scientists, Trends and Perspectives in Agriculture, no. 23. Copenhagen, Danmark, 26. juni - 29. juni 2007.

Møller, H.B., Nielsen, L. & Christensen, T. Unpublished. Biogas production from different types of biomass and grass species from meadows. Unpublished manuscript.

Nassar, A.M., Harfuch, L., Bachion, L.C., Moreira, M.R., 2011. Biofuels and land-use changes: searching for the top model. Interface Focus 1: 224-232.

Naturskyddsföreningen, 2011. Policy - Skogen. Stockholm, Naturskyddsföreningen: 30.

Naturvårdsverket, 2006. Helträdsutnyttjande – konsekvenser för klimat och biologisk mångfald. Rapport 5562, May 2006. (In Swedish.)

Naylor, R. L., A. J. Liska, M. B. Burke, W. P. Falcon, J. C. Gaskell, S. D. Rozelle, and K. G. Cassman. 2008. The ripple effect: biofuels, food security, and the environment. The Environment 49(9):30-43.

Ng, F. and M. A. Aksoy. 2008. Food price increases and net food importing countries: lessons from the recent past. Agricultural Economics 39:443-452.

Nieminen, J.K., Räisänen, M. & Haimi, J. 2012. Spot mounding and granulated wood ash increase inorganic N availability and alter key components of the soil food web in clear-cut Norway spruce forests. Forest Ecology and Management 263:24-30.

Nihlgård, B. 1972. Plant Biomass, Primary Production and Distribution of Chemical Elements in a Beech and a Planted Spruce Forest in South Sweden. Oikos 23: 69-81.

Nilsson, S.I., Berggren, D. and Westling, O., 1998. Retention of deposited NH4-N and NO3-N in coniferous forest ecosystems in southern Sweden. Scandinavian Journal of Forest Research 13, 339-341.

Nilsson, J. and Timm, B., 1983. Environmental effects of wood and peat combustion. Summary and conclusions. SNV PM 1844. SNV (Swedish Environmental Protection Agency), Solna.

Nohrstedt, H.-Ö. 2001. Response of coniferous forest ecosystems on mineral soils to nutrient additions: a review of Swedish experiences. Scandinavian Journal of Forest Research 16: 555-573.

Nordén, B., Götmark, F., Tönnberg, M. & Ryberg, M. 2004. Dead wood in semi-natural temperate broadleaved woodland: contribution of coarse and fine dead wood, attached dead wood and stumps. Forest Ecology and Management 194:235-248.

Nordén, U. 1994. Influence of Tree Species on Acidification and Mineral Pools in Deciduous Forest Soils in South Sweden. Water, Air, and Soil Pollution 76: 363-381.

Nordin, A., Lundmark, T., Grip, H., Nilsson, M. and Ericson, L., 2009a. Miljöanalys av behovsanpassad gödsling på skogsmark. Slutrapport P30639-1. Energimyndigheten, Eskilstuna.

Nordin, A., Bergström, A.-K., Granberg, G., Grip, H., Gustafsson, D., Gärdenäs, A., Hyvönen-Olsson, R., Jansson, P-E., Laudon, H., Nilsson, M.B., Svensson, M. and Öquist, M., 2009b. effekter av ett intensivare skogsbruk på skogslandskapets mark, vatten och växthusgaser. Faktaunderlag till MINT-utredningen. SLU, Uppsala.

Nord-Larsen, T. 2002. Stand and Site Productivity Response Following Whole-Tree Harvesting in Early Thinnings of Norway Spruce (Picea Abies (L.) Karst.). Biomass and Bioenergy 23(1):1-12.Norström S., 2010. Soil and stream water chemistry in a boreal catchment – interactions, influences of dissolved organic matter and effects of wood ash application. Doctoral Thesis 99, Mittuniversitetet, Sundsvall.

O´Connor, D., 2011. Issues with GTAP iLUC Modelling, Presentation at Seminar: Indirect land-use change related to biofuels and bioliquids: what option would you chose? Organised by Copa-Cogeca, Brussels 3rd May 2011.

OECD. 2008. Biofuel Support Policies: An Economic Assessment. Organisation for Economic Co-operation and Development (OECD), Paris.

Olsson, B.A. & Kellner O. 2002. Effects of soil acidification and liming on ground flora establishment after clear-felling of Norway spruce in Sweden. Forest Ecology and Management 158:127–139.

Olsson B.A., Staaf H., Lundkvist H., Bengtsson J. and Rosen K., 1996. Carbon and nitrogen in coniferous forest soils after clearfelling and harvest of different intensities. Forest Ecology and Management 82, 19-32.

Olsson, M., 1996. Långsigtiga näringsbalanser vid uttag av skogsbränsle. Kungliga Skogs- och Lantbruksadaemiens tidskrift 13, 37-44.

Olsson, M., Andersson P., Lennartsson T., Lenoir L., Mattsson L. & Palme U. 2011. Land management – effects on greenhouse gas emissions, biodiversity and water. Unpublished report, SLU, Uppsala.

Olsson, R. 2011. Hugga eller Skydda? Boreala skogar i klimatperspektiv. Air Pollution and Climate Series. Stockholm och Göteborg, Svenska Naturskyddsföreningen och Världsnaturfonden WWF i samarbete med Luftförorenings- och klimatsekretariatet (AirClim): 64.

Otte, A,. Ginzler, O., Waldbardt, R. & Simmering, D. 2008. The common pasture of the nature reserve "Kanzelstein bei Eibach" (Lahn-Dill Kreis, Hessen, Germany): Change and state of a biotope complex of a pre-industrial cultural landscape. Tuexenia 28:151-184.

Peeters, A. 2009. Importance, evolution, environmental impact and future challenges of grasslands and grassland-based systems in Europe. Grassland Science 55(3):113-125.

Penner, J.E., Dickinson, R.E., O'Neill, C.A. 1992. Effects of Aerosol from Biomass Burning on the Global Radiation Budget. Science, Vol. 256, pp 1432-1434.

Perala, D.A. and Alban, D.H., 1982. Biomass, nutrient distribution and litterfall in Populus, Pinus and Picea stands on two different soils in Minnesota. Plant and Soil 64, 177-192.

Pettersson, F. and Högbom, L., 2004. Long-term growth effects following forest nitrogen fertilization in Pinus sylvestris and Picea abies stands in Sweden. Scand J For Res 19, 339-347.

Pielke, R.A., Marland, G., Betts, R.A., Chase, T.N., Eastman, J.L., Niles, J.O., Running, S.W., 2002. The influence of land-use change and landscape dynamics on the climate system: relevance to climate-change policy beyond the radiative effect of greenhouse gases. Philosophical Transactions of the Royal Society of London. Series A: Mathematical, Physical and Engineering Sciences 360, pp 1705-1719.

Piesse, J. and C. Thritle. 2009. Three bubbles and a panic: An explanatory review of recent food commodity price events. Food Policy 34:119-129.

Pitman R., 2006. Wood ash use in forestry – a review of the environmental impacts. Forestry 79(5), 563-588.

Plantureux, S., Peeters, A. & McCracken, D. 2005. Biodiversity in intensive grasslands – effect of management, improvement and challenges. Agronomy Research 3:153–164.

Plevin, R.J., O´Hare, M., Jones, A.D., Torn, M.S., Gibbs, H.K., 2010. Greenhouse Gas Emissions from Biofuels. Indirect Land-use change Are Uncertain but May Be Much Greater than Previously Estimated. Environmental Science & Technology 44, 8015-8021.

Popp, A., J. P. Dietrich, H. Lotze-Campen, D. Klein, N. Bauer, M. Krause, T. Beringer, D. Gerten, and O. Edenhofer. 2011. The economic potential of bioenergy for climate change mitigation with special attention given to implications for the land system. Environmental Research Letters 6(3):034017.

Powers R.F., Scott D.A., Sanchez F.G., Voldseth R.A., Page-Dumroese D., Elioff J.D. and Stone D.M., 2005. The North American long-term soil productivity experiment: findings from the first decade of research. Forest Ecology and Management 220, 31-50.

Prochnow, A., Heiermann, M., Plöchl, M., Linke, B., Idler, C., Amon, T. & Hobbs, P.J. 2009a. Bioenergy from permanent grassland – A review: 1. Biogas. Bioresource Technology 100:4931–4944.

Prochnow, A., Heiermann, M., Plöchl, M., Amon, T. & Hobbs, P.J. 2009b. Bioenergy from permanent grassland – A review: 2. Combustion. Bioresource Technology 100:4945–4954.

Prins, A. G., B. Eickhout, M. Banse, H. v. Meijl, W. Rienks, and G. Woltjer. 2011. Global impacts of European agricultural and biofuel policies. Ecology and society 16(1):art49.

Prins, A.G., Stehfest, E., Overmars, K.P., Ros, J., 2010. Are models suitable for determining iLUC factors?, PBL, Netherlands Environmental Assessment Agency.

Rajagopal, D., S. E. Sexton, G. Hochman, D. Roland-Holst, and D. Zilberman. 2009. Model estimates food-versus-biofuel trade-off. California Agriculture 63(4):199-201

Rajagopal, D., S. E. Sexton, D. Roland-Holst, and D. Zilberman. 2007. Challenge of biofuel: filling the tank without emptying the stomach? Environmental Research Letters 2(4):044004.

Ranger, J., Marques, R., Colin-Belgrand, M., Flammang, N. and Gelhaye, D., 1995. The dynamics of biomass and nutrient accumulation in a Douglas-fir (Pseudotsuga menziesii Franco) stand studies using a chronosequence approach. Forest Ecology and Management 72, 167-183.

Raulund-Rasmussen, K., Borggaard, O.K., Hansen, H.C.B. and Olsson, M., 1998. Effect of natural organic soil solutes on weathering rates of soil minerals. Eur. J. Soil Sci. 49, 397-406.

Ravallion, M. 1990. Rural Welfare Effects of Food Price Changes under Induced Wage Responses: Theory and Evidence for Bangladesh. Oxford Economic Papers 42(3):574-585.

Reddersen J., Nordvig K. & Jensen T.S. 2005. Energipil som habitat for småpattedyr (Mammalia: Rodentia, Insectivora) I et blandet dansk landbrugslandskab. Flora og Fauna 111:81–89.

Renssen, S. v. 2011. "A biofuel conundrum." Nature and climate change Volume 1, November 2011.

Repo, A., M. Tuomi, *et al*. 2010. "Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues." GCB Bioenergy 3: 107-115.

Ring, E., Lövgren, L. Nohrstedt, H-Ö. and Jansson, G., 1999. Ash fertilization in a clearcut and in a Scots pine stand in central Sweden. Skogforsk Report No 2, 50 pp.

Ring, E., Högbom, L. and Nohrstedt, H-Ö., 2001. Effects of brash removal after clear felling on soil and soil-solution chemistry and field-layer biomass in an experimental nitrogen gradient. The Scientific World Journal 1, 457–466.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature 461: 472-475.

Roberts, M. J. and W. Schlenker. 2009. World Supply and Demand of Food Commodity Calories. American Journal of Agricultural Economics 91(5):1235-1242.

Robertson, G.P., Hamilton, G. K., Del grosso, S. J. and Parton, W. J., 2011. The biogeochemistry of bioenergy landscapes: carbon, nitrogen, and water considerations, Ecological Applications, 21(4):1055–1067.

Roesch, C., Skarka, J., Raab, K. & Stelzer, V. 2009. Energy production from grassland – assessing the sustainability of different process chains under German conditions. Biomass and Bioenergy 33(4):689-700.

Rosegrant, M. W. 2008. Biofuels and Grain Prices: Impacts and Policy Responses. Page 4 Testimony 48, U.S. Senate Committee on Homeland Security and Governmental Affairs, May 7, 2008, Washington, D.C.

Rosegrant, M. W., T. Zhu, S. Msangi, and T. Sulser. 2008. Global Scenarios for Biofuels: Impacts and Implications. Review of Agricultural Economics 30(3):495-505.

Rosen, S., S. Shapouri, K. Quanbeck, and B. Meade. 2008. Food security assessment, 2007. Economic Research Servive, United States Department of Agriculture, Washington, D.C. 54.

Rosenqvist, H; Roos, A; Ling, E; *et al*. 2000. Willow growers in Sweden. Biomass & Bioenergy 18 (2) 137-145.

Rowe, R.L, Street, N.R. Taylor, G., 2009. Identifying potential environmental impacts of large-scale deployment of dedicated bioenergy crops in the UK, Renewable and Sustainable Energy Reviews 13(1): 271-290.

RTRS, 2010. "RTRS EU RED Compliance Requirements for Producers Version 2.0." ProForest for the RTRS Executive Board and Biofuel WG.

Rudolphi, J. & Gustafsson, L. 2005. Effects of forest-fuel harvesting on the amount of dead wood on clear-cuts. Scandinavian Journal of Forest Research 20:235-242.

Rumpf, S., Ludwig, B. and Mindrup, M., 2001. Effect of wood ash on soil chemistry of a pine stand in Northern Germany. J. Plant Nutr. Soil Sci. 164, 569-575.

Rydberg, D. & Falck, J. 1996. Den mångsidiga skottskogen. FaktaSkog 8/1996, SLU.

Rytter, L., Johansson, T., Karacic, A., Weih, M. 2011. Orienterande studie om ett svenskt forskningsprogram för poppel. Arbetsrapport / Skogforsk vol. 733.

Rytter, RM. 2001. Biomass production and allocation, including fine-root turnover, and annual N uptake in lysimeter-grown basket willows. For Ecol Manage 140:177-192.

Rytter, R.M., 2012. The potential of willow and poplar plantations as carbon sinks in Sweden. Biomass and Bioenergy 36: p. 86-95.

Röser, D., Asikainen, A., Raulund-Rasmussen, K., and Stupak, I., 2008. Sustainable Use of Forest Biomass for Energy: A Synthesis With Focus on the Baltic And Nordic Region. Springer verlag, ISBN 1402050534, 9781402050534, 262 pp.

Röös, E., Sundberg, C., Hansson, P-A. 2011. Uncertainties in the Carbon Footprint of Refined Wheat Products: A Case Study on Swedish Pasta. International Journal of LCA. 2011, Volume 16, Number 4, Pages 338-350.

Saarsalmi, A., Mälkönen, E. and Piirainen, S., 2001. Effects of wood ash fertilisation on forest soil chemical properties. Silva Fenn. 35: 355-367.

Sahlin, M. 2011. Under the Cover of the Swedish Forestry Model. Stockholm, Swedish Society for Nature Conservation: 36.

Sahlin, M. 2010. Cutting the Edge - the loss of Natural Forests in Sweden. Stockholm, Swedish Society for Nature Conservation: 32.

Salin, P. 2012. Skogen vi ärvde - ”Skogen är vårt levebröd”. DN.se. Dagens Nyheters nätupplaga: 2.

Sanchez, F.G., Scott, D.A. and Ludovici, K.H., 2006. Negligible effects of severe organic matter removal and soil compaction on loblolly pine growth over 10 years. Forest Ecology and Management 227, 145-154.

Sandberg, M. 2012. ”De är inte ute så mycket”. Den bostadsnära naturkontaktens betydelse och utrymme i storstadsbarns vardagsliv. PhD Monography, Institutionen för kulturgeografi och ekonomisk geografi. Handelshögskolan vid Göteborgs Universitet. 243.

Sandell, K. and M. Svenning. 2011. Allemansrätten och dess framtid. Stockholm, Naturvårdsverket: 227.

Sanderson, M. A. and Adler P. R., 2008. Perennial Forages as Second Generation Bioenergy Crops, Int. J. Mol. Sci. 9: 768-788.

Scarlat, N., Dallemand, J.-F. 2011. "Recent developments of biofuels/bioenergy sustainability certification: A global overview." European Commission, Joint Research Centre, Institute for Energy, published in Energy Policy 39(3): 1630 – 1646.

SCB, 2012. Jordbruksverket, Naturvårdsverket och LRF. Hållbarhet i svenskt jordbruk 2012. ISBN 978-91-618-1570-8.

SCB, 2008. "Markanvändningen i Sverige." Femte utgåvan, Statistiska Centralbyrån.

Schaaf W., 1995. Effects of Mg(Oh)(2)-Fertilization on Nutrient Cycling in A Heavily Damaged Norway Spruce Ecosystem (Ne Bavaria/Frg). Plant and Soil 169, 505-511.

Schaber-Schoor, G. 2009. Forestal production of wooden energy considering sustainability of deadwood and biodiversity. Forst und Holz 64(2):14-17.

Schmidhuber, J. 2007. Biofuels: An emerging threat to Europe’s Food Security? Impact of an Increased biomass use on agricultural markets, prices and food security: A longer-term perspective. Policy Paper 27, Notre Europe. 40.

Schulz U., Brauner O. & Gruss H. 2009. Animal diversity on short-rotation coppices – a review. Landbauforschung Volkenrode 59(3):171-181.

Schulze, E.D. *et al*. 2009. Importance of methane and nitrous oxide for Europe´s terrestrial greenhouse-gas balance. Nature GeoScience 2:842-850.

Scott Wilson, 2009. Friends of the Earth - Developing a spreadsheet model for the calculation of the emissions from indirect land-use change (iLUC) as a result of biofuel production-explanatory note.

Scott, D.A. and Dean, T.J., 2006. Energy trade-offs between intensive biomass utilization, site productivity loss, and ameliorative treatments in loblolly pine plantations. Biomass & Bioenergy 30, 1001-1010.

Searchinger, T., Heimlich, R., Houghton, R.A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.-H., 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. Science 319: 1238-1240.

Selman, P. 2010. Learning to Love the Landscapes of Carbon-Neutrality. Landscape Research, 35:2, 157-171.

Semere, T. & Slater, F.M. 2007a. Ground flora, small mammal and bird species diversity in miscanthus (Miscanthus x giganteus) and reed canary-grass (Phalaris arundinacea) fields. Biomass and Bioenergy 31(1): 20–29.

Semere, T. & Slater F.M. 2007b. Invertebrate populations in miscanthus (Miscanthus x giganteus) and reed canary-grass (Phalaris arundinacea) fields. Biomass and Bioenergy 31(1):30-39.

Shane, S., & Venkataraman, S. 2000. The promise of entrepreneurship as a field of research. Academy of Management Review, 25(1): 217-226.

Shapouri, S., S. Rosen, B. Meade, and F. Gale. 2009. Food security assessment, 2008-2009. GFA-20, Economic Research Service, United States Department of Agriculture, Washington, D.C. 58.

Shine, K.P., Fuglestvedt, J.S., Hailemariam, K., Stuber, N. 2005. Alternatives to the global warming potential for comparing climate impacts of emissions of greenhouse gases. Climatic Change, 68(3), 281-302.Shurpali, N. J., Hyvönen, N., Huttunen, J. T., Clement, R., Reichestein, M., Nykänen, H., Biasi, C., and Martikainen, P. J. 2009. Cultivation of perennial grass for bioenergy use on a boreal organic soil – carbon sink or source? Global Change Biol. Bioenergy, 1, 35–50.

Sikström U., Jacobson S., Johansson U., Kukkola M., Saarsalmi A., Holt-Hansen K., 2009. Långtidseffekter på skogsproduktion efter askåterföring och kalkning – en pilotstudie med preliminära resultat. Slutrapport Q6-660, Värmeforsk.

Silfverberg, K. and Huikari, O., 1985. Tuhkalannoitus metsäojitetuilla turvemailla. Folia For. 633: 1-25.

Silfverberg, K. and Issakainen, J., 1987. Tuhkan määrän ja laadun vaikutus neulasten ravinnepitoisuuksiin ja painoon rämemänniköissä. Metsäntutkimuslaitoksen tiedonantoja 271, 1-25.

Silfverberg, K. and Moilanen, M., 2000. Environmental consequences of recycling wood ash to forests. In: Högbom L. and Nohrstedt H.-Ö., (Eds.) Extended abstracts from the SNS (Nordic Forest Research Coordination Committee) workshop at Grimsö, Sweden, 22-25 May 2000. SkogForsk Report 2, SkogForsk, Uppsala, pp 25-28.

Simard, D.G., Fyles, J.W., Paré, D. and Nguyen, T., 2001. Impacts of clearcut harvesting and wildfire on soil nutrient status in the Quebec boreal forest. Canadian Journal of Soil Science 81, 229-237.

Simcock, R.C., Parfitt, R.L., Skinner, M.F., Dando, J. and Graham, J.D., 2006. The effects of soil compaction and fertilizer application on the establishment and growth of Pinus radiata. Canadian Journal of Forest Research 36, 1077-1086.

SJV, 2008a. Ängs-och betesmarker – en genomgång av tillgänglig statistic. Rapport 2008:30, Jordbruksverket, Jönköping

SJV, 2008b. Kartläggning av mark som tagits ur produktion. Rapport 2008:7, Jordbruksverket, Jönköping. ISSN 1102-3007.

SJV, 2012. Jordbruksmarkens användning 2011. Statistiska Meddelande JOSM1201, Jordbruksverket, Jönköping.

Skogsstyrelsen, 2011. Skogsstatistisk årsbok 2011. J.-O. Loman. Jönköping, Skogsstyrelsen: 386.

SKSFS, 2007. Skogsstyrelsens allmänna råd till ledning för hänsyn enligt 30 § skogsvårdslagen (1979:429) vid användning av kvävegödselmedel på skogsmark. ISSN 0347-5212. Beslutade den 17 april 2007.

Skärbäck E. & Becht P. 2005. Landscape perspective on energy forests. Biomass and Bioenergy 28: 151-159.

SLU, 2011. Skogsdata 2011. Umeå, Institutionen för skoglig resurshushållning, Statens Lantbruks Universitet (SLU) och Sveriges Officiella Statistik: 128.

SLU, 2010. Halvtidsutvärdering av landsbygdsprogram för Sverige 2007-2013. Landsbygdsdepartementet Jo 10.013. http://www.sweden.gov.se/sb/d/13746/a/155799

SLU, 2007. Slututvärdering av Miljö- och landsbygdsprogrammet 2000-2006 - vad fick vi för pengarna? Utredning, Dnr SLU ua 12-3269/07.

Smith C.T., Lowe A.T., Skinner M.F., Beets P.N., Schoenholtz S.H. and Fang S., 2000. Response of radiata pine forests to residue management and fertilisation across a fertility gradient in New Zealand. Forest Ecology and Management 138, 203-223.

Sohi, S. P., Krull E., Lopez-Capel, E., and Bol, R. 2010. A Review of Biochar and Its Use and Function in Soil, Advances in Agronomy, Volume 105 , ISSN 0065-2113.

Sonntag-Öström, E., M. Nordin, *et al*. 2011. "Can the boreal forest be used for rehabilitation and recovery from stress-related exhaustion? A pilot study." Scandinavian Journal of Forest Research 26(3): 245-256.

Soo, T., Tullus, A., Tullus, H. & Roosaluste, E. 2009. Floristic diversity responses in young hybrid aspen plantations to land-use history and site preparation treatments. Forest Ecology and Management 257:858-867.

Soussana, J.F., Tallec, T., Blanfort, V. 2010. Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. Animal 4:334-350

Spracklen, D.V., Bonn, B., Carslaw, K.S., 2008. Boreal forests, aerosols and the impacts on clouds and climate. Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences 366: 4613-4626.

Staaf, H. & Olsson, B.A., 1994. Effects of slash removal and stump harvesting on soil water chemistry in a clearcutting in SW Sweden. Scandinavian Journal of Forest Research 9, 305-310.

Staun, H. & Klitgaard, O.L. 2000. Staevningsskove på Fyn og Langeland – oversigt og status. Dansk Skovbrugs Tidsskrift 85(2):53-98.

Steenari, B.M., Karlsson, L.G. and Lindqvist, O., 1999. Evaluation of the leaching characteristics of wood ash and the influence of ash agglomeration. Biomass and Bioenergy 16, 119-136.

Strengbom, J. 2009. Kärlväxter, lavar och mossor. Pages 97-127 in: Gustafsson L., Dahlberg A., Green M., Henningsson S., Hägerhäll C., Larsson A., Lindelöw Å., Lindhagen A., Lundh G., Ode Å., Strengbom J., Ranius T., Sandström J., Svensson R. & Widenfalk O. 2009. Konsekvenser för kulturarv, friluftsliv, landskapsbild och biologisk mångfald. Faktaunderlag till MINT-utredningen. Report, SLU, Uppsala.

Strengbom, J. & Nordin, A. 2008. Commercial forest fertilization causes long-term residual effects in ground vegetation of boreal forests. Forest Ecology and Management 256:2175–2181.

Stridsman, M. 2012. Skogen vi ärvde - ”Nej, Zaremba, varken lagen eller vi kräver kalhyggen”. DN.se. Dagens Nyheters nätupplaga: 2.

Strömgren, M. 2009. Kort- och långsiktiga effekter på markkolet vid stubbrytning. Slutrapport av projekt 30474-1. Energimyndigheten, Eskilstuna.

Stupak, I., Asikainen A., Jonsell M., Karltun E., Lunnan A., Mizaraite D., Pasanen K., Parn H., Raulund-Rasmussen K., Roser D., Schroeder M., Varnagiryte I., Vilkriste L., Callesen I., Clarke N., Gaitnieks T., Ingerslev M., Mandre M., Ozolincius R., Saarsalmi A., Armolaitis K., Helmisaari H.S., Indriksons A., Kairiukstis L., Katzensteiner K., Kukkola A., Ots K., Ravn H.P. & Tamminen P. 2007a. Sustainable utilisation of forest biomass for energy-possibilities and problems: policy, legislation, certification, and recommendations and guidelines in the Nordic, Baltic, and other European countries. Biomass and Bioenergy 31:666–684.

Stupak, I., Kukkola M., Varnagiryte-Kabasinskiene I., 2007b. Biomass and nutrient removals in Norway spruce, Scots pine and birch for whole rotations: modelling the effects of harvesting intensity, site productivity, target diameters, and nutrient concentration levels in biomass. Manuscript in Stupak, I. 2007. Biomass and nutrient removals in sustainable forest fuel harvesting. A site and stand specific approach to risk evaluation and governance. Ph.D.-dissertation, Forest & Landscape, Faculty of Life Sciences, University of Copenhagen.

Ståhl, P. 2009. Produktionshöjande åtgärder. Jönköping, Skogsstyrelsen: 77.

Swedbio, 2009. Biofuels – Potential and Challenges for Developing Countries. Facts from SwedBio no 3, Swedish Biodiversity Centre, Uppsala.

TEEB, 2010. The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB., The Economics of Ecosystems and Biodiversity (TEEB) study - United Nations Environmental Programme (UNEP): 39. Available at: www.teebweb.org/Portals/25/TEEB%20Synthesis/TEEB_SynthReport_09_2010_online.pdf

Thiele, R. 2003. Price Incentives, Non-price Factors and Agricultural Production in Sub-Saharan Africa: A Cointegration Analysis. African Development Review 15(2-3):425-438.

Timilsina, G. R., J. C. Beghin, D. Van der Mensbrugghe, and S. Mevel. 2010. The Impacts of Biofuel Targets on Land-Use Change and Food Supply: A Global CGE Assessment. Policy Research Working Paper 5513, World Bank, Washington, D.C. 71.

Tipper, R., Hutchison, C., Brander, M., 2009. A Practical Approach for Policies to Address GHG Emissions from Indirect Land-use change Associated with Biofuels, Ecometrica and Greenergy. Technical Paper - TP-080212-A.

Tiwari, S. and H. Zaman. 2010. The Impact of Economic Shocks on Global Undernourishment. World Bank, Policy Research papers. Washington, D.C.

Trostle, R. 2008. Global Agricultural Supply and Demand: Factors Contributing to the Recent Increase in Food Commodity Prices. WRS-0801, US Department of Agriculture (USDA).

Tuttle C.L., Golden M.S. and Meldahl R.S., 1985. Soil surface removal and herbicide treatment: effects on soil properties and loblolly pine early growth. Soil Science Society American Journal 49, 1558-1562.

Tyner, W. E. 2010. The integration of energy and agricultural markets. Agricultural Economics 41:193-201.

Tyner, W. E. and F. Taheripour. 2008. Policy options for integrated energy and agricultural markets. Review of Agricultural Economics 30(3):387-396.

Tyner, W.E., Taheripour, F., Zhuang, Q., Birur, D., Baldos, U., 2010. Land-use changes and Consequent CO2 Emissions due to US Corn Ethanol Production: A Comprehensive Analysis., July 2010. Department of Agricultural Economics, Purdue University.

UNCTAD, 2008. United Nations Conference on Trade and Development, Making Certification Work for Sustainable Development: The Case of Biofuels, New York and Geneva.

UNECE and FAO. 2011. The European Forest Sector Outlook Study II 2010-2030. Rome, United Nations Economic Commission for Europe (UNECE) and Food and Agriculture Organization of the United Nations (FAO): 111.

USEPA, 2010. Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis. Assessment and Standards Division Office of Transportation and Air Quality U.S. Environmental Protection Agency. EPA-420-R-10-006 February 2010.

Van Breemen N., Mulder J. and Driscoll C.T., 1983. Acidification and alkalinization of soils. Plant and Soil 75, 283-308.

Van Stappen, F. Brose, I., Schenkel, Y., 2011. "Direct and indirect land-use changes issues in European sustainability initiatives: State-of-the-art, open issues and future developments." Biomass and bioenergy 35: 4824 - 4834.

Vejre H., Ingerslev M. and Raulund-Rasmussen K., 2001. Fertilization of Danish Forests: A Review of Experiments. Scandinavian Journal of Forest Research 16(6), 502-513.

Voytenko and Peck. 2012. Organisational frameworks for straw-based energy systems in Sweden and Denmark. Biomass and Bioenergy, 38 (1) 34-48.

Waldenström, R. Ferguson, C. Sundberg, P. Tidåker, E. Westholm, A. Åkerskog. 2011. Institutional barriers for rural development; the example of bioenergy from farmland. XXIV European Society for Rural Sociology Congress, 22-25 August 2011, Chania, Greece.

Wall A., 2008. Effect of removal of logging residue on nutrient leaching and nutrient pools in the soil after clearcutting in a Norway spruce stand. For. Ecol. Manage. 256, 1372-1383.

Walmsley J.D., Jones D.L., Reynolds B., Price M.H. and Healey J.R., 2009. Whole tree harvesting can reduce second rotation forest productivity. Forest Ecology and Management 257, 1104-1111.

Wang J., LeDoux C.B., Edwards P. and Jones M., 2005. Soil bulk density changes caused by mechanized harvesting: A case study in central Appalachia. Forest Products Journal 55 (11), 37-40.

Watmough S.A. and Dillon P.J., 2003. Base cation and nitrogen budgets for a mixed hardwood catchment in South-Central Ontario. Ecosystems 6, 675-693.

Watt M.S., Coker G., Clinton P.W., Davis M.R., Parfitt R., Simcock R., Garrett L., Payn T., Richardson B. and Dunningham A., 2005. Defining sustainability of plantation forests through identification of site quality indicators influencing productivity - A national view for New Zealand. Forest Ecology and Management 216, 51-63.

Weih M. 2006. Energiskogsodling på åkermark – möjligheter för biologisk mångfald och kulturmiljö i ett landskapsperspektiv. Rapport till Naturvårtdsverket Dnr: 802-114-04.

Weih M., Karacic A., Munkert H., Verwijst T. & Diekmann M. 2003. Influence of young poplar stands on floristic diversity in agricultural landscapes (Sweden). Basic and Applied Ecology 4:149-156.

Wert S. and Thomas B.R., 1981. Effects of skid roads on diameter, height, height and volume growth in Douglas-fir. Soil Science Society American Journal 45, 629-632.

Westling O., Örlander G. & Andersson I., 2004. Effekter av askåterföring till granplanteringar med ristäkt. Rapport B1552. IVL, Stockholm.

White E.H., 1974. Whole-tree Harvesting Depletes Soil Nutrients. Canadian Journal of Forest Research 4, 530-535.

White House, 2012. National Bioeconomy Blueprint. Washington, The White House: 48.

Widell, L.M., 2009. Ekonomiska modeller för jordbrukssektorn – en redogörelse för deras uppbyggnad, funktion och förmåga att beskriva svenska förhållanden, Jordbruksverket Rapport 2009:23.

Wiggins, S., E. Fioretti, J. Keane, Y. Khwaja, S. McDonald, S. Levy, and C. S. Srinivasan. 2008. Review of the indirect effects of biofuels: Economic benefits and food insecurity. London.

Winsley, P., 2007. Biochar and bioenergy production for climate change mitigation, New Zealand Science Review Vol 64 (1)

Wodon, Q. T., C. Tsimpo, P. Backiny-Yetna, G. Joseph, F. Adoho, and H. Coulombe. 2008. Potential Impact of Higher Food Prices on Poverty: Summary Estimates for a Dozen West and Central African Countries. World Bank, Policy Research Papers, Washington, D.C. 31.

Woolf, D., 2008. Biochar as a soil amendment: A review of the environmental applications, Available at: http://orgprints.org/13268/1/Biochar_as_a_soil_amendment_-_a_review.pdf

World Bank, 2012. Prices (Pink Sheet): Monthly world prices of commodities and indices. Washington, D.C.

Wretenberg J., Lindström Å., Svensson S., Thierfelder T. & Pärt T. 2006. Population trends of farmland birds in Sweden and England: similar trends but different pattern in agricultural intensification. Journal of Applied Ecology 43:1110-1120.

WWF, 2011. Hållbar Energi. 100% förnybart på naturens villkor. Stockholm, Stiftelsen Världsnaturfonden (WWF): 20.

WWF, 2009. Remissvar ”Skog utan gräns?”. http://www.wwf.se/source.php/1268670/090907%20-%20Remissvar%20Skog%20utan%20grns.pdf

Xiong SJ and Kätterer T, 2010. Carbon-allocation dynamics in reed canary grass as affected by soil type and fertilization rates in northern Sweden ACTA AGRICULTURAE SCANDINAVICA SECTION B-SOIL AND PLANT SCIENCE 60 (1) 24-32.

Yeh, S., Witcover, J., 2010. Indirect Land-Use Change from Biofuels: Recent Developments in Modeling and Policy Landscapes, International Food & Agricultural Trade Policy Council. Downloaded March 2011 www.agritrade.org/events/documents/Yeh_Witcover_iLUCfromBiofuels.pdf.

Zabrowski D., Chambreau D., Rotramel N and Thies W.G., 2008. Long-term effects of stump removal to control root-rot on forest soil bulk density, soil carbon and nitrogen content. Forest Ecology and Management 255, 720-727.

Zanchi, G., N. Pena, *et al*. 2011. "Is woody bioenergy carbon neutral? A comparative assessment of emissions from consumption of woody bioenergy and fossil fuel." GCB Bioenergy: n/a-n/a.

Zaremba, M. 2012a. Maciej Zaremba: I rörelsernas Sverige har skogen ingen talan. DN.se. Dagens Nyheters nätupplaga: 2.

Zaremba, M. 2012b. Skogen vi ärvde - Chatt med Maciej Zaremba om skogen. DN.se. Dagens Nyheters nätupplaga: 4.

Zaremba, M. 2012c. Skogen vi ärvde - Lagen är en rökridå. DN.se. Dagens Nyheters nätupplaga: 5.

Zaremba, M. 2012d. Skogen vi ärvde - Motorsågsmassakern. Det finns alternativ, men lagen kräver kalhyggen. DN.se. Dagens Nyheters nätupplaga: 5.

Zaremba, M. 2012e. Skogen vi ärvde - Skogens maffia. Så tog de makten över träden. DN.se. Dagens Nyheters nätupplaga: 5.

Zaremba, M. 2012f. Skogen vi ärvde - Skönheten och odjuren. Så drivs människan ut ur skogen. DN.se. Dagens Nyheters nätupplaga: 5.

Zaremba, M. 2012g. Skogen vi ärvde - Sveriges nya miljonprogram. DN.se. Dagens Nyheters nätupplaga: 5.

Zetterberg, L. 2011. Instruments for reaching climate objectives. Focusing on the time aspects of bio-energy and allocation rules in the European Union's Emission Trading System. Doctoral Thesis. University of Gothenburg, Department of Earth Sciences.

Zetterberg T., Hellsten S., Belyazid S., Karlsson P.E. & Akselsson C. 2006. Regionala förutsättningar och miljörisker till följd av skogsmarksgödsling vid olika scenarier för skogsskötsel och kvävedeposition – modellerade effekter på kväveupplagring, biomassa, markkemi. IVL Svenska Miljöinstitutet AB, rapport B1691.

Zetterberg T. and Olsson B., 2011. Long-term effects of clear-cutting and biomass removal on soil water chemistry at three coniferous sites in Sweden. IVL report B1959.

Zezza, A., B. Davis, C. Azzarri, K. Covarrubias, L. Tasciotti, and G. Anriquez. 2009. The Impact of Rising Food Prices on the Poor. International Association of Agricultural Economists Conference, Beijing.

Åström, K. and J. Terstad. 2012. Skogen vi ärvde - ”Därför behövs en ny skogspolitik”. DN.se. Dagens Nyheters nätupplaga: 2.

Åström M. 2006. Aspects of heterogeneity: effects of clear-cutting and post-harvest extraction of bioenergy on plants in boreal forests. Dissertation Umeå University, Dept. of Ecology and Environmental Science.