Environmental assessments and Swedish consumption of biofuels

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Review of Swedish biofuel research and aggregated life cycle assessment of Swedish biofuel consumption 2000-2013

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Abstract

This thesis assesses the potential environmental impacts for Swedish biofuel consumption and evaluates the environmental scope in Swedish biofuel research. The biofuel consumption is portrayed with life cycle assessment methodology and presents the emissions occurring from different feedstock and fuel production regions. The outcome of the biofuel consumption is compared with a fossil fuel scenario in order to put the emissions of biofuels in broader context. A systematic literature review is conducted on the published Swedish environmental system analysis on biofuels, to quantitatively interpret assessed environmental indicators and methods. The findings are that while global warming potential emissions decreased, local environmental impact potentials increases drastically. Most of the environmental burdens from the consumption have occurred in the EU and South America. Results from the systematic literature review shows on a dominating scope towards global warming potential and energy performance of the fuels. The justifications for the low consideration of other impact categories are often without motivation. Implications occurring from the limited scope are discussed in terms of potential problem shifting and its correspondence to Swedish environmental policy.

The biofuel policies from the European Union and the Swedish Government are insufficient regarding mitigation of e.g. eutrophication, acidification and toxicity. Impacts from land use change and indirect land use change are also highly important to include in the assessments.

Sammanfattning

I denna masteruppstats bedöms de potentiella miljöeffekterna för Svensk biodrivmedelskonsumtion och analysomfattningen i den Svensk biodrivmedelsforskningen. Sveriges konsumtion av biodrivmedel är miljömässigt bedömd genom livscykelanalys, där emissioner från råvaruframställning och produktion presenteras över uppkomstområde åren 2000-2013. De olika påverkansindikatorerna jämförs med miljöpåverkan från förbränning av fossila drivmedel med samma energiinnehåll. En systematisk litteraturundersökning genomförs över alla svenskpublicerade vetenskapliga artiklar i syfte att kvantitativt bedöma omfattningen av påverkanskategorier. Resultaten visar att samtidigt som växthusgaserna minskat har andra påverkanskategorier ökat dramatiskt. Den bedömda miljöpåverkan till följd av biodrivmedelskonsumtionen skedde till största delen i Europeiska och Sydamerikanska länder. Resultatet från den systematiska litteraturundersökningen visar ett övervägande i miljöpåverkansfokus på växthusgasutsläpp och energiprestanda i forskningen. Valet av påverkanskategorier är ofta undermåligt beskrivet och motivering till de studerade påverkansområdena saknas i majoriteten av rapporterna. Konsekvenserna av denna snäva miljöpåverkanssyn är diskuterad i förhållande till problemförflyttning och svensk miljöpolicy. Biodrivmedels strategier från Europeiska Unionen och den svenska regeringen är bristfälliga för att minska t. ex. övergödning, försurning och toxicitet. Påverkan från förändrad landanvändning och indirekt förändrad landanvändning är även de viktiga att medta i bedömningarna.

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List of abbreviations

AD	Abiotic depletion
AG	Agriculture
DLUC	Direct land use change
DME	Dimetylether
EP	Eutrophication potential
ESA	Environmental system analysis
FAME	Fatty acid methyl eter
FT	Fischer tropsch
GWP	Global warming potential
HVO	Hydrotreated vegetable oil
IEA	International energy agency
ILUC	Indirect land use change
IPCC	International panel for climate change
ISO	International standardisation organisation
LBG	Light burning gas
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
POCP	Photocehmical oxidant formation potential
PROD	Production
RME	Rapeseed methyl eter
SEA	Strategic environmental assessment/Swedish energy agency
SNG	Syntetic natural gas
TP	Toxicity potential
WD	Water depletion

1. Introduction

Sweden is heavily dependent on fossil energy resources imported from a few oil extracting countries. The transportation fleet in Sweden consists of rail, road, flight and marine traffic, of which the road transports accounted for 94% of the energy use in 2013 (SEA, 2014d). The road transportation sector is mainly supplied by imported fossil fuels. The renewable part share is increasing from small amounts in 2000 for a 10.5% share of the consumed transport fuels in 2013 (Ibid.). Sweden has the largest part of biofuels in the transportation sector in the world (Ibid.). These renewable fuels are liquid or gaseous, produced from biomass crops or bio waste. The consumed fuels are ethanol, Fatty Acid Methyl Ester (FAME), Hydro treated Vegetable Oils (HVO), liquid and gaseous biogas, Ethyl Tertiary Butyl Ether (ETBE) and Dimethyl Ether (DME) (SEA, 2014a).

An agreement is made by the Swedish Government in proposition 2008/09:162 to phase out the fossil fuels (SOU, 2013). The reasons for the shift towards renewable transport fuels is mainly depending on fuel dependency, fuel security, possible domestic technical developments, health and environmental aspects (SOU, 2013). There is a policy pressure to increase the biofuels, both from EU's renewable directive (Directive 2009/70/EC, 2015) EU's fuel directive (Directive 98/70/EG) and Sweden's parliament(SOU, 2013). Sweden has ambitious goals for the vehicle fleet to be independent of fossil fuels in 2030 and greenhouse gas (GHG) neutral, sustainable and resource efficient in 2050(SOU, 2013).

According to the Swedish law (SFS:2010:598) the GHG emissions from the biofuels has to decrease with at least 35% in 2015 and 50% in 2017, compared to fossil fuels. The law includes a sustainability criterion. This criterion is mainly covering aspects of carbon emissions with few demands on other impacts (SFS2010:598). The GHG reduction has to be assessed with life cycle terminology.

Life cycle assessment (LCA) is an analytical tool used to assess the impacts of a product or service over its life cycle(Baumann and Tillman, 2004). The life cycle assessment method is required both in the European renewable directive and by its application in Swedish law SFS:2010:598 on GHG emissions (SOU, 2013). There has been a rapid increase of LCAs on biofuels (Cherubini and Strømman, 2011; van der Voet, Lifset and Luo, 2010).

Sweden's environmental burden is shifting from a domestically view to follow the trade patterns of products linked to environmental impacts in other countries (SEPA, 2013). The environmental impacts from the producer country need to be included for a complete assessment of the products environmental life cycle burden (Ibid.). These impacts come from many different sources and countries due to import and export of products (Brolinson et al., 2010). The impacts occur in different stages and areas in the products´ lifecycle (Baumann and Tillman, 2004), and a life cycle approach is necessary (SEPA, 2013). In 2010 the Swedish parliament reformulated the environmental policy to include emissions occurring outside Sweden:

"The overall goal of Swedish environmental policy is to hand over to the next generation a society in which the major environmental problems in Sweden has been solved, without increasing environmental and health problems outside Sweden's boarders" (SEPA, 2013)

The Swedish Environmental Protection Agency (SEPA) started a project to quantify Sweden's emissions in a project called consumption based indicators. The SEPAs project´s aim is to portray the airborne emissions over time to see trends, not to examine exact levels. Other areas of interest to assess are chemicals, land use, water use and biodiversity. The reason for the SEPAs focus on airborne emissions is because there are already robust methods for those impacts, further development is needed for the other emission areas (SEPA, 2013). The Swedish environmental protection agency also wants to further portray where the abroad emissions are released, and investigate the areas of protection further.

International studies on life cycle assessment have indicated that impact categories investigated in biofuel research are often limited to GWP and energy balances (Cherubini and Strømman, 2011; van der Voet, Lifset and Luo, 2010; Bai, Luo and Van Der Voet, 2010). Even though the assessments are limited they are still used for policy making (van der Voet, Lifset and Luo, 2010). A limited scope in LCA can mislead policy making and cause problem shifting to other emissions (Laurent, Olsen and Hauschild, 2012).

Currently, there are no reviews over the conducted environmental assessments of biofuels in Swedish research. A review is necessary to determine the environmental scope in the Swedish biofuel research. There are either no studies presenting the regional impacts for the consumed fuels, according to the SEPAs consumption patterns. A consumption based LCA would provide important knowledge on the emissions of biofuels. Such assessment would both give information about levels of emissions and the areas where they are released. Therefore the research questions are:

- How has the environmental impacts been assessed in the Swedish biofuel research?

- Where and to which magnitude has environmental impacts occurred due to Swedish biofuel consumption over time?

Aim and objectives

The aim of this thesis is twofold. First, to analyze the environmental scope in Swedish biofuel research. Second, to portray the environmental impacts connected to biofuel consumption over time. The objectives are:

I) To analyze the environmental impacts and methods assessed in biofuel environmental impact research in Sweden 2000-2014

II) To account for the impacts caused from biofuel consumption in Sweden from 2000-2013 and compare with fossil fuel scenario

III) Outline the potential implications of limiting the impact categories in biofuel environmental assessments

This thesis is divided in two subparts to answer the aims. At first a review is done in order to determine the scope in Swedish biofuel research. The second subpart is providing an investigation closely connected to the environmental protection agency's consumption indicator project, to increase the knowledge about biofuels overall impact over time. To put the connected impacts of the biofuels in a bigger context a comparison with fossil fuels will be made.

The thesis is in synthesis with two F3 Centre projects: Carbon Vision? *Reviewing Environmental Analyses of Biofuel Production in Sweden* and *Accumulated Impacts from Increased Biofuel Consumption in Sweden.* F3 center is a platform for biofuel research in Sweden.

1.2 Scope of the study

Only biofuels for Swedish transportation purposes are investigated. The assessment considers only environmental impacts, not social or economic.

The systematic literature review considers studies written or co-written with at least one Swedish author. The articles have been published the years between 2000-2014 in Scopus, Web of Science, Springer or F3 Fuels database. The included articles have a life cycle perspective and are ESAs (environmental system analysis) or sustainability reports. All invested reports have at least included and assessed one environmental impact. Economic and social aspects are outside the scope.

The LCA is done on Ethanol, FAME, HVO and biogas. DME and methanol are not included. The consumed amounts of the excluded fuels are small and datasets are unavailable.

Land use, direct and indirect land use change are not assessed. Those environmental categories are excluded since there are not any reliable Life Cycle Inventory data for all biofuels available.

2. Background

The background gives information about fuel types, previous research, biofuel policies and life cycle assessment.

2.1 Transportation biofuels

Transportation biofuels are liquid or gaseous fuels produced from biomass, waste and by products and used for transportation. In Sweden the fuels are consumed in form of ethanol, FAME, HVO, biogas both liquid and gaseous, ETBE and DME (SEA, 2013b). The total amount of bioenergy components in transportation fuels is 10.5% in 2013 (SEA, 2014b).

Ethanol

Ethanol is produced from grains or cellulose in a fermentation process. Different pre-treatment processes are needed depending on feedstock. Ethanol from sugar crops like sugar beets, demands a low pre-treatment compared to lignocellulosic, that requires a hydrolysis step to break the chemical bindings. The hydrolysis is achieved by acids or enzymes. Globally, ethanol is by far the most common biofuel, representing 90% of the global biofuel volume. The domestic ethanol accounts for around a fourth of the domestic consumption, other countries of origin are primary Lithuania, France and Brazil. Since 2004, ethanol has been used as low blend in gasoline fuel and later as E85 for light transport vehicles and ED95 in heavier transport vehicles (F3-Centre, 2015a).

Biogas

Biogas can be produced from different feedstock like organic waste, residues from agriculture and energy crops. The production can be done separately or in co-production with for example ethanol. The formation occurs during anaerobic digestion. There are roughly 240 digestion plants in Sweden. Around half of those industries produce vehicle gas. The gained energy and efficiency of those depends on the plants performance. The efficiency is generally better for the plants handling waste sludge, municipal organic waste and industrial waste than for production from manure and energy crops. This is often due to the greater energy input for harvesting and maintaining the agriculture phase. The production requires thermal energy to hygenisize the biogas. All biogas is upgraded to increase the methane content and remove sulfur compounds. Biogas is a mixture of nature gas and biogas where the amount biogas has been around 60-63%. The biogas is produced from a variety of feedstock, mainly from rest products and waste on different scale and production sites. Biogas or the bio methane[1] can be used in passenger cars and busses. (SOU, 2013).

Methanol

Methanol is produced in small quantities and mostly used in chemical industry. Methanol has good properties as biofuel, only small adjustments are required for use in conventional combustion engines. Today the development is shifting from the fossil based methanol to biomass and agricultural waste as feedstock. Another development is to produce methanol from gasification of black liquor, which is a byproduct from the pulp and paper industry. Methanol can be used as a biofuel or used as a component for production of DME, gasoline or biodiesel. It’s the simplest form of alcohol produced via synthesis gas, which consists of hydrogen and carbon monoxide. (F3-Centre, 2015e)

[1] Biomethane is a upgraded form of raw biogas with methane content of 96-98%

Di methyl ether (DME)

Traditionally dimethyl ether(DME) is produced from coal or natural gas. Development is occurring in production of DME from Black liquor. The bio-based DME is otherwise from dehydration of methanol. DME is an interesting option since it has high cetane and low octane number. Therefore it is suitable for replacement of fossil diesel. Diesel engines have to be converted with a new injection system before DME can be driven on (F3-Centre, 2015b).

Fatty acid methyl ester (FAME)

FAME (fatty acid methyl esters), commonly referred to as biodiesel. Soybean, rapeseed, sunflower, jatropha, corn and animal fats can be used for production, giving different chemical compositions. In Sweden FAME is produced from rapeseed, the fuel is called RME (rapeseed methyl ester). The production process includes transesterification to obtain FAME by separation of the glycerin by an alcohol with a catalyst. FAME is used in vehicles either blended as component in fossil diesel or as a complete substitute used in heavy trucks (F3-Centre, 2015c).

Hydrotreated vegetable oils (HVO)

Most compounds including fatty acids can be used for HVO production. Common feedstock are many vegetable oils and fats like rapeseed, soy bean and tall oil. The feedstock passes a pretreatment and hydrotreatment under high pressure where the triglycerides react to remove oxygen. The consumption of HVO in Sweden has increased with 100% between 2011 and 2012. HVO has properties making it appropriate for substitution of diesel. (F3-Centre, 2015d).

Fischer tropsch and hydogen

Fischer Tropsch is an advanced biofuel. All types of biomass can act as feedstock, both agricultural and forest biomass. The biomass is gasified and thereby transformed to synthesis gas consisting of hydrogen and carbon monoxide. The gas is, via a catalyst, turned into hydrocarbon fractions and further converted to vehicle fuels. Bio hydrogen can also be produced by gasification of biogas, and is further transformed into hydrogen with a water gas shifter. The fuel can be either blended with conventional fuels or employed as it is. (Elobio, 2015).

2.2 Policies

There are two fundamental stakeholders in the policies of biofuels: the Swedish government and the EU parliament (Holmgren, 2012). The Swedish government has ambitious environmental goals for a future transport sector. In 2030 the vehicle fleet should be independent on fossil fuels and in 2050 there should not be any emissions of GHG to the atmosphere (SOU, 2013). The main purpose from the Swedish government is to stimulate the technological development of more environmental friendly fuels (Ibid.).

2.2.1 Goals and directives from the European Union

The EU has several directives concerning biofuels: The Renewable Energy Directive 2009/28/EG (RED), the Fuel Quality Directive 2009/30/EC (FQD), the Sustainability Directive (Holmgren, 2012). EU is propagating for a system change in the 2009/28/EG directive (RED). The RED aims for a common framework to promote the development for renewable energy utilization, limit greenhouse gas emissions and to promote a cleaner transportation system. Sweden has a binding aim towards the RED target of a 49% share of energy from renewable sources in 2020 (European Parliament, 2015). In the transportation sector the minimum share should be 10% (ibid.). Some biofuels produced from waste, rest products or products from cellulose may be double counted towards the target. HVO and biogas is fulfilling the double counting criteria (Ibid.).

The fuel quality directive 2009/30/EC has the aim to ensure a single market for the fuels and to limit the environmental impact when using the fuels. The directive sets limits for the oxygen content, which for example limit the blending in gasoline to maximum 10% ethanol. The maximum blending rate of biodiesel in diesel is 7%. This directive has GHG reduction targets to promote better GHG performance on biofuels and reduce fossil fuels (European Parliament, 2009a).

To reach a transition towards less fossil dependent fuels without sub optimizing phases, there is a sustainability directive for the biofuels in directive 2009/28/EG (European Parliament, 2009). The sustainability directive accounts for a life cycle perspective, to not shift impacts from one life cycle into another. The directive is implemented in the Swedish law, Law (2010:598). By this law all producers are responsible to report their produced amounts and sustainability declaration to the Energy Department (SEA, 2014c). The sustainability criteria, Law (2010:598) include a GHG reduction with at least 35%, 50% reduction in 2017, compared to fossil fuels. Furthermore, the law 2010:598 prohibits growing crops or forest on nature forests, grasslands with high biodiversity, wetlands and on protected areas. The biofuels shall also be reported and be traceable (Swedish constitution, 2010).

2.2.2 Goals and instruments from the Swedish government

In the proposition 2008/09:162 the Swedish government accounts for the long term priority for the transportation sector. It states that Sweden in 2030 should have a vehicle fleet independent on fossil fuels (Prop. 2008/09:162, 2008). The definition from the fossil free investigation, SOU 2013:84 is that a fossil independent vehicle system is “A vehicle transport system that is mainly driven with biofuels or electricity”. Proposition 2008/09:162 also states in the long term vision in 2050 Sweden should have a resource efficient and sustainable energy supply without any emissions of GHGs.

Several instruments are applied to promote the biofuels. The Biofuels are exempted from carbon and energy taxes. There are mandatory blending rates in the fossil gasoline and diesel. The government strives to increase those blending rates. Different premiums and tax subsidies have been applied to new biofuel driven cars. In 2006 a “pump law” was implemented. The pump law required all tank stations of a certain size to provide at least one renewable fuel option (Holmgren, 2012).

2.3 Environmental system analysis tools (ESA-tools)

Environmental system analysis is defined by Wageningen University as “Environmental System Analysis is a quantitative and multidisciplinary research field aimed at analyzing, interpreting, simulating and communicating complex environmental problems from different perspectives” (Wageningen University, 2015).

Systems are complex by its nature, they consists of subsystems with no clear links between them. By not focusing on separate parts, a holistic approach is taken (Moberg, 2006). By studying through system analysis, aiming at a wider perspective, sub-optimization can be reduced or avoided. Environmental system analysis considers tools and methods for assessing the impacts of human-made systems in connection to environment using a system perspective(Moberg, 2006). The different ESA-tools, their objects and coverage of impacts can be seen in figure 1. ESA tools can be analytical or procedural, analytical tools are providing quantitative or qualitative data for a better technical description and understanding of the system. Procedural tools are aiming at improving procedures for decision-making (Ibid.).

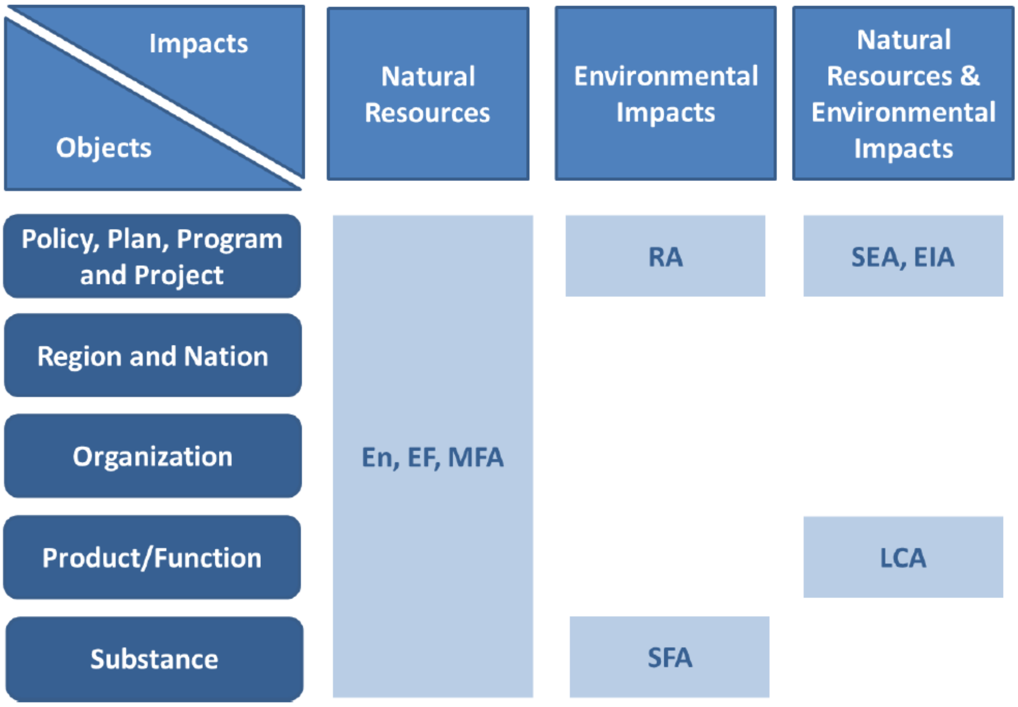

Figure 1 Environmental system analysis tools and their objects studied. Adapted from Moberg (2010). En-energy analysis, EF-Ecological footprint, MFA-Material flow analysis, RA-Risk assessment, SFA-substance analysis, SEA-Strategic environmental assessment, EIA- Environmental impact assessment, LCA- Life cycle assessment

2.4 Life cycle assessment (LCA)

Life cycle assessment is an analytical ESA-tool to determine environmental consequences on a product or service through it whole lifetime (Guniee', 2002). The environmental burdens are all type of emissions or interference with the environment caused by the product or service (Ibid.). The scope is cradle to grave which implies a holistic approach to involve impacts that already occurred and will occur. The life cycle covers all stages from the extraction of resources, through the production of materials, product parts and the product itself, and the use of the product to the management after it is discarded (Ibid.). The total impact is summarized from the impacts of each subsystem that together make up the whole product system. It is a comprehensive assessment of environmental interventions as it considers all attributes or aspects of the natural environment, human health and resources (ISO, 2006). The LCA is defined through the performing procedure:

"LCA is a technique for assessing the environmental aspects and potential impacts associated with a product by:

- Compiling an inventory of relevant inputs and outputs of a product system;
- Evaluating the potential environmental impacts associated with those inputs and outputs;
- Interpreting the results of the inventory analysis and impact assessment phases in relation to the objectives of study"

(ISO, 1997)

2.4.1 Phases of life cycle assessment

The method for LCA is structured along a worldwide consensus framework formed to the basis of a numbers of ISO standards. The framework divides the LCA into four phases: Goal and scope definition, Inventory analysis, impact assessment and interpretation (Guniee', 2002). In figure 2 the general framework for LCA is shown. The goal and scope definition defines the goal, system boundaries and functional unit. The inventory analysis is the phase for identification and quantification of data. In the impact assessment, the substances from the LCI are evaluated on magnitude and significance of the potential environmental impacts. The interpretation is where inputs from all three previous steps are assessed (Moberg, 2006).

Goal and scope definition

In the goal and scope definition the product to be studied and the purpose of the study are decided on (Baumann and Tillman, 2004). The standard ISO 1440 (1997) stresses that the goal and scope of an LCA study must be clearly defined and consistent with the intended application. Questions to be answered from the LCA should be stated. The context for the LCA shall be explained. A clear functional unit is given. A functional unit is defined as the measure of the function of the studied system, it provides a reference to which the inputs and outputs can be related. The functional unit shall be related to the function or service in quantitative terms (Guniee', 2002). Other choices to be decided on and expressed in the goal and scope definition are the system boundaries, types of environmental impacts considered, level of detail in the project and requirements of data (Guniee', 2002)

Life cycle inventory analysis

Life cycle inventory (LCI) is the phase when the system is build according to the goal and scope definition. The system is a model of material and energy in correspondence to the boundaries, only environmental relevant flows are considered. The data for all outputs and inputs of substances, emissions and waste for the activities is gathered (Guniee', 2002).

Life cycle impact assessment

Life cycle impact assessment (LCIA) aims to describe and indicate the environmental impacts of the environmental loads collected in the LCI. One of the purposes of the LCIA is to turn the inventory results into more environmentally relevant information. Another purpose is to aggregate the information from the LCI in fewer parameters (Baumann and Tillman, 2004).

The LCIA consists of classification and characterization. In the classification the inventory analysis parameters are sorted to each impact category. The characterization is the calculation where the relative contribution of the emissions and resources are calculated to each type of impact category. Usually, the classification and characterization is done automatically by software with pre-defined characterization methods (Baumann and Tillman, 2004).

Interpretation

The process of assessing results in order to draw conclusions is called interpretation. Evaluation of the robustness of conclusions is also part of the interpretation phase. The conclusions must be in correlation to the goal and scope of the LCA. Significant issues are identified in connection with the other LCA phases (Baumann and Tillman, 2004). This phase is the most subjective part in the LCA since decisions are to be made on the relative importance of the impact categories (ISO, 2006).

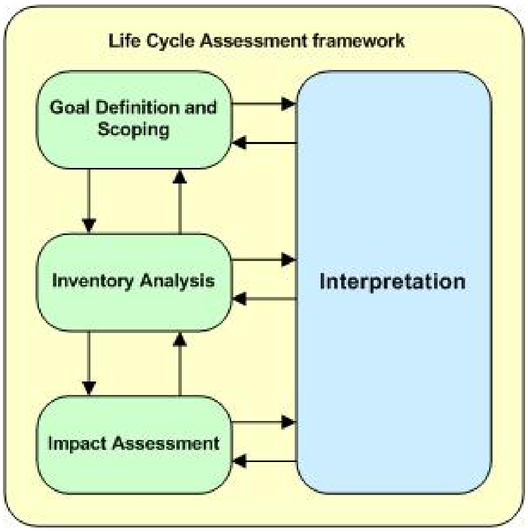

Figure 1 Phases of LCA (Baumann and Tillman 2004)

2.4.2 Impact categories

There are many impact categories that can be studied from a LCA. Three areas for protection are commonly assessed: resource use, human health and ecological consequences (Baumann and Tillman, 2004).

SETAC is a non-profit global organization working for solving environmental problems (SETAC, n.d). The SETAC recommendation from 1996 gives the following recommendation on environmental indicators: Abiotic resources, biotic resources, land, global warming, depletion of stratospheric ozone, human toxicological impacts, ecotoxicological impacts, photo-oxidant formation, acidification and eutrophication (Haes et al., 1999).

Resources

Resources can be divided into abiotic and biotic resources (Baumann and Tillman, 2004). Abiotic resources are those considered as ”non-living” resources such as iron ore and crude oil. Biotic resources are “living” i.e. those with a biological character. Examples are forests, animals and plants. There is also a distinction made whether resources are deposits, funds and flows. Deposits are resources that are not regenerated within a lifetime, also called non-renewable resources. Funds can be regenerated within a lifetime and flows are resources that are constantly regenerated (Haes et al., 1999).

Land use

Three land use indicators are considered: the actual use of land and changes in land use (Baumann and Tillman, 2004). Land use also covers the extent to which land use land and land transformation leads to changes in biodiversity and to life support functions. More about biofuels and land use change in section 2.6.1.

Global warming

Characterisation of greenhouse gases is based on the extent which they enhance the radiative forcing in the atmosphere. The potential contribution of a substance to climate change is expressed as its global warming potential (GWP). GHGs' have different life spans in the atmosphere and can be calculated for different time horizons (Baumann and Tillman, 2004).

Ozone depletion

The ozone layer is an essential substance in the upper atmosphere. Atmospheric ozone screens out 99% of the ultraviolet radiation from the sun. Depletion of the ozone occurs as various chlorinated and substances react with the ozone. Examples of catalysts that cause destruction of the ozone are H, OH, NO, CL and Br. The ozone depletion potentials (ODPs) were developed by the World Meteorological Organisation. (Baumann and Tillman, 2004)

Toxicity

Toxicity is complicated to assess. The reason for the complexity is that toxicity includes many impacts and almost all substances (Baumann and Tillman, 2004). The impacts are different, some substances causes neurological damage others carcinogenic, mutagenic etc. Toxins have often high ability to disperse and it's often hard to know the end up sink. Toxicity is often divided into ecotoxicity and human toxicity. Furthermore, ecotoxicity can be divided in aquatic toxicity and terrestrial toxicity. And then the aquatic toxicity into freshwater and marine toxicity, and there is also freshwater and marine sediment toxicity (Ibid.).

The main difference between the many characterisation approaches depends on the effect and extent to which the fate of the substance that has been included. The development of accurate methods is limited by the availability to fate and background data (Ibid.).

Photo-oxidant formation

Photo-oxidants are secondary reactions pollutants formed in lower atmosphere with appearance of NO_x and hydrocarbons in presence of sunlight. These substances are known as smog, causing health problems and damage in vegetation (Baumann and Tillman, 2004). The damage on agriculture crops from photo-oxidant formation is substantial. Important photo-oxidants are ozone, peroxyacetyl nitrate, hydrogen peroxide and aledehydes. Photo-oxidant formation is both a local and regional problem (Baumann and Tillman, 2004).

Acidification

Examples of acidification effects are fish mortality, leaching of toxic metals, damage on forests and buildings. The major pollutants are $SO_2, NO_x, HCl\ and\ NH_3$ (Sea and Water Agency, 2014). In Sweden, the acidification emissions from forestry are increasing rapidly, due to the removal of roots. The roots are important for mitigation of acidification since they bind biologic compounds (Ibid.). All acidifying pollutants form H^+ ions. The acidification potential (AP) is defined as the number of H^+ ions produced per kg substance relative to SO_2 (Baumann and Tillman, 2004).

Eutrophication

The occurrence of excessively high levels of nutrients lead to shifts in species composition and increased biological productivity is called eutrophication (Baumann and Tillman, 2004). The most implicated nutrition are nitrogen and phosphorous. Both terrestrial and aquatic ecosystems are affected. A eutrophied aquatic system has decreased oxygen levels and an increased biomass formation. Eutrophication potential is measured as PO_4^{3-} eqv. (Baumann and Tillman, 2004).

2.4.3 Allocation

Allocation is a necessary when several products or functions share the same process or processes (Baumann and Tillman, 2004). The environmental burdens connected to the same processes sometimes needs to be allocated between the products or functions. There are three basic cases when allocation occurs: Multi output allocation, multi input allocation and open loop recycling (Ibid.)

Multi output allocation is applied when a process result in several products. When a process has many inputs of products a multi-input allocation problem occurs. Open loop considers recycling of a product into others (Ibid.).

The choice of allocation is dependent on the LCA type: accounting or consequential (see next section). The ISO standard does not make difference on different LCA types but is recommending an order of preference:

1. Whenever possible allocation should be avoided by a) increased level of detail of the model. b) system expansion.
2. Where allocation can't be avoided the environmental loads should be partitioned between the system's different functions. Partitioning should reflect underlying physical relationships.
3. Where physical relationships alone cannot be established or used, allocation may be based on the other relationships between the products, such as economic value.

(ISO 14040, 1998)

2.3.4 Attributional and consequential life cycle assessment

The LCA can either be conducted attributional or consequential. The attributional method is a method to describe environmentally relevant physical flows between the life cycle being studied and its subsystems. Consequential modelling is designed to understand how changes in the system due to decisions will influence the environmental performance (Finnveden et al., 2009).

Attributional LCA uses average steady state data, consequential LCA uses marginal data. The marginal data represents data that changes over time due to market mechanisms. The allocation procedures are different for the two methods. Attributional LCA solves mostly allocation problems by physical relationships e.g. energy, mass or economic values. Consequential LCAs avoid allocation by expanding the system boundaries to include affected parts of other life cycles. Consequential LCAs are generally more complex and more sensitive to assumptions used in the modelling (Finnveden et al., 2009).

2.4.4 Selection of impact categories

An important step in the LCA process is where the gathered information form the inventory analysis is transformed and interpreted to environmental impacts: the characterization. The impact categories, indicators and characterization factors have to be specified in the scope. The selection should be justified as it may have an influence of the results. If only one impact category is used the study must not be designated as LCA according to the standard (Klöpffer and Grahl, 2014). At the same time the ISO 14044 does not state which impact categories that have to be included, it is a decision to be made by the performers. The selection shall be according to the scope definition and the data in the LCI must therefore comply with the impact assessment.

Mandatory elements according to ISO 14040 and 14044:

- Selection of impact categories, category indicators and characterization models
- Assignment of LCI results (classification)
- Calculation of category indicators result (characterization)

ISO 14044 does not recommend any impact categories but refers to ISO14047 that gives recommendations. The recommendation is not as strong as a literature research to get the current development and state of indicators. Indicators and categorization models must have been accepted by an international agreement or accepted of an international board. The EU commission has planned to require a binding list of impact categories, but is not in action jet. PEF (product environmental footprint) is an example of such method (Klöpffer and Grahl, 2014).

The classification and characterization is often calculated by a LCIA-method in the software. The ready-made methods have information about the harm of the substances and aggregate those into characterization with factors. All methods have their own calculation models and therefor influencing the results (Baumann and Tillman, 2004).

2.4.5 Strengths and weaknesses of life cycle assessment

An important quality of LCAs highlighted in the Handbook by Gunieé (2002) is the assessments benefit of a cradle to grave perspective. This perspective is important to avoid “problem shifting”. In this case “problem shifting” means that an impact can be moved from a life cycle to another. LCAs can be used by governments to evaluate and making strategic decisions in policies for consumption and life styles. The limitations of LCA are its width; if all emissions and impacts from a product over its lifetime are studied there will be simplifications. The tool can´t address local impacts fully. The time dependency is also a limited factor; all emissions are calculated under steady state conditions in attributional LCAs and does not account for indirect changes and market mechanisms (Baumann and Tillman, 2004).

2.6 Life cycle assessments of biofuels

This section provides information on previous studies of life cycle assessments and biofuels, and land use change in biofuel context.

Most of the studies covering environmental performance of biofuels are based on LCA practice. The hits of a search in Scopus database for biofuels shows on a rapid increase of LCAs (see Figure 3). This graph indicates the rapid increase in research of LCA and biofuels.

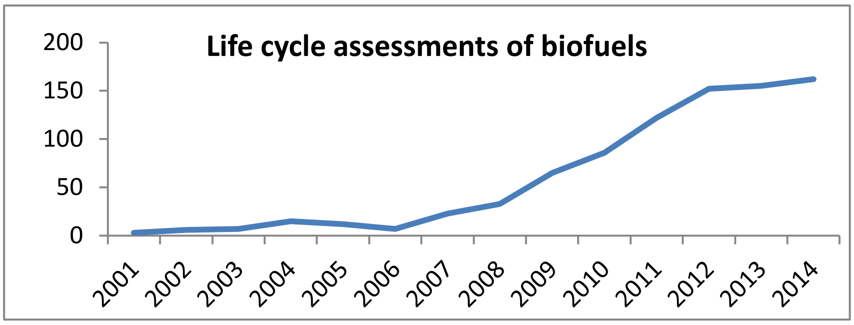

Figure 3: Hits of LCA and biofuels in Scopus

2.6.1 Previous reviews of life cycle assessments and biofuels

Many of the studies of biofuels are connected to the agriculture phase, this makes that many studies put lot of emphasis on changed land and the emissions from the cultivation (Börjesson, Lundgren, Ahlgren and Nystrom, 2013). There is one previous biofuel review over LCA reports for biofuels consumed in Sweden and their environmental impacts performed by (Linné, 2007). This article was done to investigate which LCA reports that have been committed for the different biofuels the last years to invest methods, data and transparency. The study aimed to find deficiencies in the LCAs for further assessments. Linné (2007) gives no comparison between the studies, since the research question was to examine the reliability of the LCAs. The conclusions drawn of this study were that most of the assessments only include GHG emissions and energy balances.

The allocation methods can influence the results to a large extent. More updated LCAs is needed for higher reliability. The most relevant articles in reviewing of biofuels on the international arena were Larson (2006), investigating correlations between GHG-emissions, land use with emphasis on the performance of the fuel types, Cecilé et al. (2011), gives a comprehensive review on biofuels, land use, state of the art and climate change mitigation. Rocha et al. (2014) has done a LCA based on META studies for Brazilian biofuels. Van der Voet et al. (2010) has studied the convergence and divergence in LCAs' for policy making and Cherubini and Strömman (2011) reviewed the bioenergy production, assessed the methodological issues and indirect effects.

The similarity of those reports is the aim of investigation on how clean the biofuel energy performs in greenhouse gas emissions terminology. These reports showed on the variance in energy input-output and variations in the sampled LCAs. Also, the Energy output per fossil fuel use is presented in all reviews. The most complete reviews were performed by Cécile et al. (2011) and Cherubini (2011) covering aspects of the global state of biofuels their possibilities and drawbacks of bigger biofuel utility.

LCA is the tool used in the reviews to assess the impacts for the whole life span. In this life cycle assessments all the reports point the dilemma with individual system boundaries, assumptions, limitations, co-production and allocation method causing different results for LCA performed on the same biofuels.

One difference in the performed META-studies is if they include land use or not, Larson (2006), Cécile (2011) and Cherubini (2011) discuss the soil degradation and soil organic carbon assumptions of land use change. The statement by those is that LCA isn't developed enough for simulating the asymmetrical and nonlinear relationships in soil organic carbon. The soil characteristic is highly site specific and dependent on former and current agronomic practices. Also, the biodiversity losses can be essential and isn't fully affected in the land use change LCA-indicators.

Rocha et al. (2014) and Cherubini (2011) are the only studies considering other impacts than GWP, energy ratios and land use for comparison between the LCAs.

Rocha et al. (2014) is conducting a LCA for the Brazilian ethanol and biodiesel compared through a META study with previous publicized LCAs. Four impact categories are assessed beside GWP: abiotic depletion potential, human toxicity potential, acidification potential and eutrophication potential. The aim of the meta-study was to give a comprehensive comparison between the two fuels.

Cherubini and Strömman (2011) invested 97 LCA studies worldwide, 47 of those were scope limited to GHG emissions and energy balances. Only 20% were assessing other airborne emissions and as few as 9% considered the land use change. Toxicity potential and fresh water usage weren't either included in the majority of the LCAs.

Even though there are relevant reviews over the LCA literature internationally few of them critically discusses the limitations of the outcomes. A LCA method should cover potential environmental consequences through the lifetime of a product or services(Baumann and Tillman, 2004). In reality the emphasis is mostly on the global warming emissions. The literature review is giving evidence that mitigation of climate change is the main driving force for biofuel research. It's a risk to delimit the impact categories and overlook concepts of vital importance in favor for energy balances and GHG savings if the assessments are used in policy making (Finnveden, 2000). All reviewers except Linné (2007) explain the need for a regulatory standard for a fair comparison between the fuels. Cherubini and Strömman claims for a regulation for the LCA to include wider scope of effects and indicators.

2.6.2 Land use change

Land use change is an important aspect for biofuels (Hansen, Cederberg and Hansson, 2013). There are different approaches to assess the impacts from land use change from biofuels. The land use change can cover different environmental consequences. There is a distinction between direct land use change, DLUC and indirect, ILUC. DLUC aims to cover the aspects occurring from a transformation of a land area from one state to another. ILUC considers the potential effects from DLUC when another area is transformed due to the DLUC. The DLUC are uncertain but can be measured, whereas ILUC can't. Land use changes causes many effects, some of them have been more studied than others. The range is from life supporting mechanisms, ecosystem services to greenhouse gasses and nutrient leakage. Thereby are the fields interconnected; soil chemistry, biodiversity and climate change are all affected by LUC. The effects from LUC are highly site specific, depending on which land is converted, which crops, management practices etc. The intensity of cultivation might affect the quality and erosion, composition and nutrient content of the soil. (Hansen, Cederberg and Hansson, 2013)

2.7 Previous studies over Swedish accumulated emissions in connection to road transports

The greenhouse gas (GHG) emissions from the transport sector are well known and portrayed over time in reports from the Swedish Energy Agency (SEA) for Sweden's fuel consumption (SEA, 2014d) and by the Swedish Energy Protection Agency (SEPA, 2015) (See Figure 4) . Other examples of Sweden's transportation GHG emissions are reported by Sweden's government (Swedish Ministry of Environment, 2013). The reporting from SEA and the Swedish Government are based on the values from the SEPA. The SEPA is responsible for the GHG reporting (SEPA, 2015; Swedish Ministry of Environment, 2013).

The road traffic stands for 95% of the GHG emissions, which is around a third of the national GHG emissions (SEPA, 2015). Since 2007 has the emissions from the road traffic started to decrease. The main part of the emissions is from personal vehicles and heavy trucks, see Figure 4. The emissions from personal cars have decreased although the car use increased overall. This emission decrease is a result of a more energy efficient car development and an increased utilization of biofuels (Ibid.).

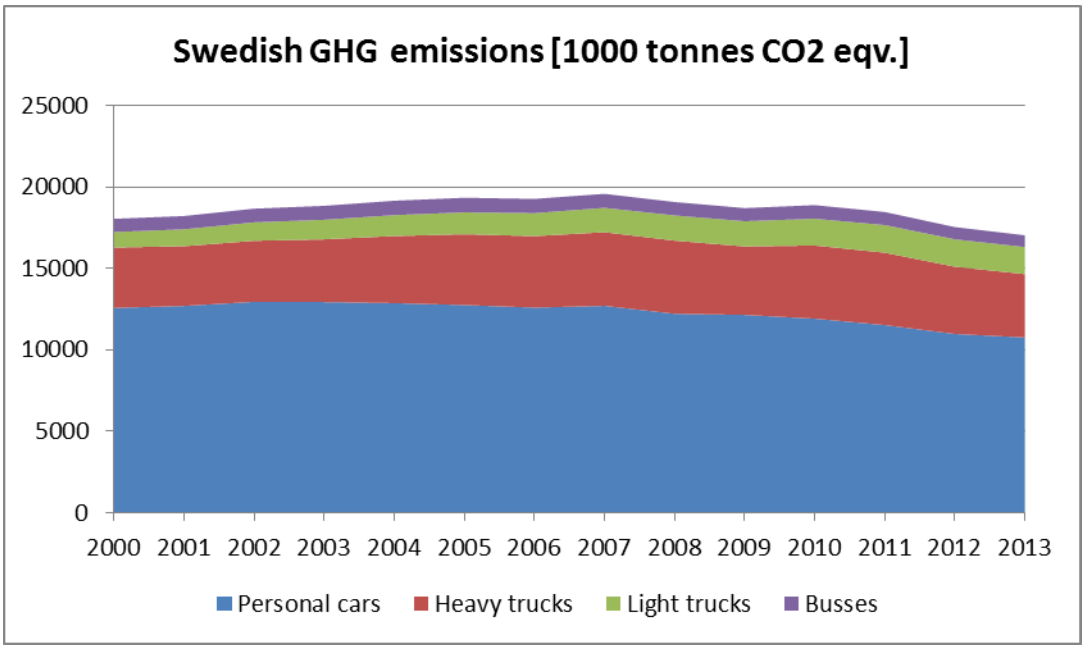

Figure 4: Sweden's accumulated emissions from transport vehicles (SEPA, 2015).

2.7.2 Other Swedish emission databases

SMED is a consortium founded in 2001 with an aim to gather data and develop competence on emissions in Sweden in collaboration with Statistics Sweden, the SEPA and IVL (The Swedish Environmental Institute). SMED has data on air emissions e.g. SO_2, NOX, $NMVOC$ and NH_3 and N and P pressures on fresh and sea water (SMED, 2014). One project portrays the particles released from the abrasion from the road traffic (Gustafsson and Jerksjö, 2011). However, there are no reports besides GHG emissions and particles that are directly linked to the road traffic.

The Swedish Protection Agency has several databases for surveillance of emissions to show on trends in the environment and an emission register (SEPA, n.d.). Example of data is levels of toxins, air emissions and eutrophication emissions. Similar to the SMED database some important industrial sectors are presented but not the road traffic with exception of the GHG emissions.

These reports have not considered where the emissions occurred in a life cycle perspective and therefore not reported according to the consumption based patterns as the Swedish environmental agency recommends. More about the concept consumption based patterns are explained in section 3.2.

3. Method

Different methods are used to fulfil the objectives and provide answers to the aims. To fulfill objective I) a meta-study is made to evaluate the outcome of Swedish ESA biofuel research. For objective II) an accumulated LCA is performed based on the yearly consumption volumes. The accumulated LCA requires a production and feedstock analysis, to collect the input data for the assessment. Objective III) is fulfilled by the outcome of the meta-study and the accumulated LCA, together with a literature study over impacts from biofuels.

3.1 Systematic literature review

A systematic literature review aims to synthesize research findings from a large number of different studies. In particular, when the intervention is to potentially be used to inform policy and practice in the field investigated (Ridley, 2008).This correlates well with the thesis aim to identify the methods and impact categories in biofuel research in Sweden, and investigate if there is some general limitations in the environmental impact assessment in the research field on Swedish biofuels.

A systematic review out of the conducted LCAs in Sweden is therefore made for comparison and to generalize findings of environmental assessment methods. A META study is the qualitative interpretation of a systematic literature review, preferred to merge the results and methods on multiple articles and investigate numerically. The META study was performed with prospectively defined methods. Since there is no standardizations for conducting reviews on LCAs and environmental assessments Zumsteg et al. (2012) recommends a strategic review method commonly used in medicine named Standardized Technique for Assessing and Reporting Reviews (STARR) published in Journal of Industrial Ecology in 2014. A complete review based on this method was performed in a similar review paper about bioenergy by Muench and Guenther (2013). The STARR method is recommended to be used on LCA studies for increasing the utility of LCA reviews, to reach appropriate generalizations of the findings, benefit the ability to update the information in future reviews and to increase transparency (Zumsteg, Cooper and Noon, 2012) . Even though its main purpose is LCAs other studies with ESA structure are supposed to suit in this review method.

In order to find all relevant articles published or written in Sweden between 2000 and 2014, a structured publication gathering was performed through Scopus, Web of Science and Springer search engines. Other publications included in this study are the reports published by F3 center. F3 center is a research platform for development of sustainable renewable transport fuels in collaboration with many technical universities and research institutes in Sweden (F3-centre, 2015). F3 has a small database with published articles with high connection to life cycle assessment and system analyses of biofuels.

3.1.1 Selection of articles

To find the relevant articles a step by step search was conducted (see Figure 5). It started broad by searching for key words with combo words published during the time period (Step A).

The key words seen in Table 1 were combined with the combination words to gather the relevant samples (see Table 1). The year was set between 2000 and 2014 and country of origin was set to Sweden. In Springer and Web of Science Sweden was used as a second key word. This search ended in 152 from Scopus, 855 from Springer and 84 from Web of Science.

In step B, valid for Springer all articles were investigated by their author information to exclude articles that not were written or Co. written by a Swedish author.

In step C, further selection was made based on exclusion criteria. This criterion was that the articles should be based upon biofuel agriculture, production or processing and their environmental performance, not only energy balances or as fuel for other processes. Only biofuels for transportation purpose were include, thereby not bioenergy assessments. Cost analyses and social assessments are outside the scope. This exclusion was handled by information in abstracts or for a few cases in introduction part of the reports.

In step D, the articles were analyzed according to methods and indicators.

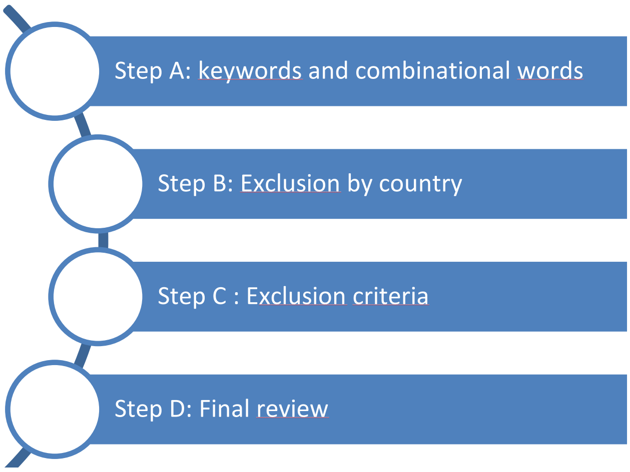

Figure 5 Search steps in meta-review

All types of biofuels for the transport sector are included. The articles studied in depth, step D, and are to be found in Appendix 1.

Three articles of interests could not be accessed (Yeh et al., 2011) “Evaluation of water use for bioenergy at different scales” and (Fingerman et al., 2011) “Impact assessment of bioenergy-water nexus” and (Sparovek et al., 2007) “Sugarcane Ethanol production in Brazil: an expansion model sensitive to socioeconomic and environmental concerns”.

Table 1: Search words

Key words									
Biofuels	Biofuel	Ethanol	Biodiesel	HVO	Biogas	Fischer Tropsch	DME	RME	Methanol
Combination words									
LCA	EIA	Life Cycle Assessment	Environmnental Impact	Life Cycle Analysis	Life Cycle Impact Assessment	SEA	Strategic Environmental Impact Assessment	SEA	

3.1.2 Assessed methods and indicators

In the studies the general documentation included feedstock, type of fuel and reference fuel. All individual reports were being documented on their scope, including limitations, system boundaries and allocation method. Environmental indicators: Global warming (GWP), abiotic depletion potential

(AD), acidification potential (AP), eutrophication potential (EP), ozone layer potential, photo chemical formation potential (POCP), land use (LU, DLUC, ILUC) and toxicity potential (TP), water depletion (WD) and impact assessment method. Also if any reason were documented for not including a complete impact assessment and which environmental performance the conclusions and recommendations were based on.

3.2 Biofuel consumption

For objective ii) to provide a yearly impact assessment for biofuels consumed in Sweden between 2000 and 2014 a raw material and production analysis is necessary. The method used have been adopted from the Swedish environmental protection agency, called consumption based patterns. The method is covering a life cycle perspective, where both the domestic and foreign production are included (SEPA, 2013; Turner, Lenzen, Wiedmann and Barrett, 2007). For reaching objective ii, multiple LCAs were performed. One for every year to assess the impact on a yearly basis to see trends and follow the impacts of imported and produced fuel products for transportation purposes. This includes a raw material and production quantification of the flows over the given time span.

Consumption is the in this thesis the annual purchases of goods and services by the private or public sector. The final domestic demand is domestically and imported produced goods and services minus export. The imported products cause emissions abroad. The emissions that are affected by production are caused in the region where the production is. For agriculture the emissions are released where the feedstock is grown. Domestic final demand is the same as domestic consumption, see figure 6 (SEPA, 2013).

The domestic final demand is the country's consumption. The consumption is equal to import and domestic production minus export (se figure 6) (SEPA, 2013). In this thesis investments and stock changes are excluded since the data is gathered from physical flows, not monetary.

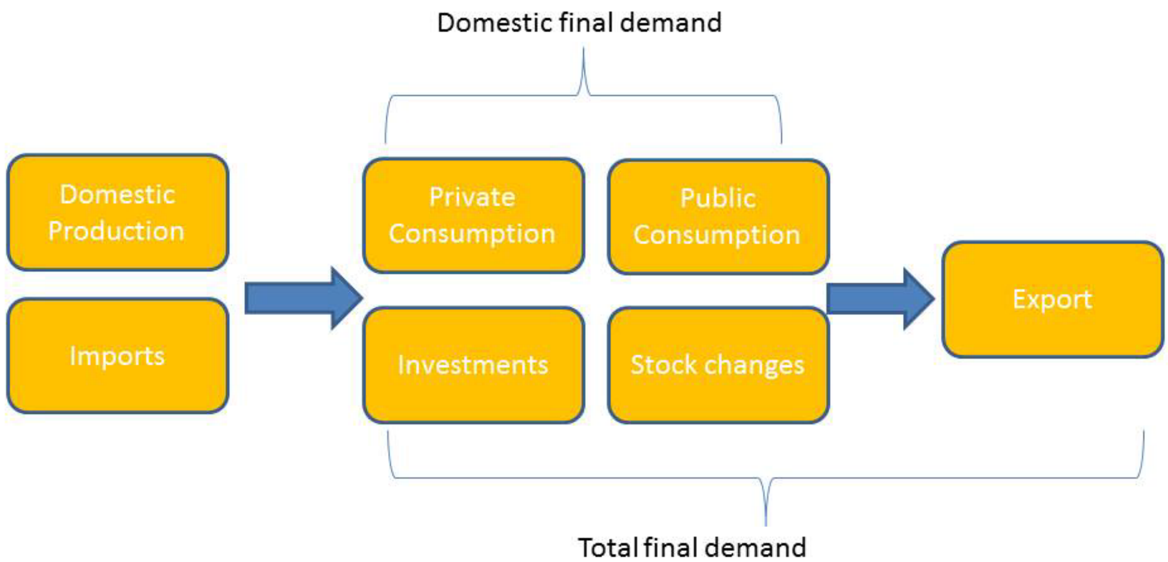

Figure 6: Economic view on consumption based pattern, adapted from (SEPA, 2013)**.**

3.3 Biofuel production and feedstock analysis

The biofuel production analysis gives the input data for the accumulated LCA. The accumulated LCA requires data for raw material, production and their provenance.

The import data was to be found in the energy department´s annually reports of energy use in the transport sector SEA (2008, 2014d), sustainable transport fuels SEA (2012, 2013b, 2014c) and market analysis for biofuels SEA (2011, 2013a, 2014a). Statistics Sweden gave the import and export of gasoline, diesel and crude oil in the database for import and export of goods (SCB, 2015). The refinery companies and the Swedish petroleum institute had some data for consumption, import and export, but this data is originally from the Swedish Energy Agency (SPBI, 2015). Statistics Sweden had shortages since the product numbers changes over the years. The product numbers are connected to a specific product. A product number can contain multiple products and the production quantity is often secrecy. The result from the flow analysis is presented in section 5.

3.3.1 Regions for feedstock and production

The regions for feedstock and production are simplified to be allocated on Sweden, rest of EU, Brazil, USA and other countries.

Sweden is trading both feedstock and biofuels from most parts of the world (SEA, 2014a). Inclusion of all those export countries in the feedstock and production analysis would be to complex. Therefore simplifications were necessary. The most influential countries of export were merged into significant countries and regions. These most influential countries were decided upon their export amounts of biofuels to Sweden obtained from Swedish Agency Sustainable fuels reports SEA (2013b, 2014c, 2012).

From 2011-2013 the ethanol feedstock come from Sweden, other EU countries (Lithuania, France, Ukraine, Poland, Romania, and Great Britain), Brazil, USA, Guatemala and other countries. The FAME feedstock come from Australia, Denmark, Lithuania, Ukraine, Poland, Sweden and other countries. The HVO feedstock has origin in Sweden, Netherlands, Germany, Indonesia, Malaysia, France, Finland and several other countries (SEA, 2012, 2013b, 2014c).

3.3.2 Assumption procedure

To make the LCA of the biofuels all life cycle steps before the Swedish vehicle market must be obtained. This includes the feedstock, agriculture and production volumes and where these steps took place. There is available information about feedstock type and the total amount imported from each region for the years 2011-2013. For the years before, only the total consumed amount per fuel is reported. To get fair controlled assumptions over the feedstock, countries of agriculture and production the assumption was divided in subparts:

A. Assume the feedstock.

B. Assume the country of agriculture or feedstock production. Consistency check for the years 2011-2013, since the total amount per country where known by Sustainable fuels reports from the energy department SEA (2014c) and by the market analysis SEA (2014a).

C. Assume the production region. Huete and Dahlbacka (2009) and the market reports SEA (2011, 2013a, 2014a) provides information on importation and production volumes of the consumed biofuels. Those values are approximations SEA (2011, 2013a, 2014a) and the values did not correspond with the reported volumes from the Swedish Energy Agency's sustainable reports (SEA, 2012, 2013b, 2014c).

3.3.3 Ethanol production and feedstock analysis

Assumptions of ethanol feedstock, the origin of the feedstock and production area are assumed according to section 3.3.2.

A) Assumptions of feedstock

Feedstock for ethanol production is presented in Appendix 2. The feedstock are based on the energy department's values for consumed ethanol per feedstock the years between 2011-2013 (SEA, 2014c) see table 2. For the years 2000-2010, no information were obtained, not from statistics Sweden and neither by the energy department. The amounts from 2000-2010 are based on assumptions and given as relative values of the total ethanol consumption (see Appendix 2 and Table 9).

Table 2 Ethanol feedstock 2011-2013 (SEA, 2014)

Ethanol feedstock m3			
	2011	**2012**	**2013**
Wheat	161000	205300	123200
Corn	115200	113900	81900
Sugarcane	47100	12500	59250
Triticale	8800	18170	52380
Barley	27700	18790	15330
Sugarbeets	6530	7465	15090
Rye	6270	2700	4076
Oat			269
Black liquor	4347	1312	93
Wine residues	7434	4035	0
Molass	5810	1331	0

B) Assumptions of agriculture and feedstock region

The assumptions for agriculture region for the feedstock are presented in Appendix 3. The total amount produced per region is balanced with the total import for the same region for 2011-2013 (SEA, 2014c).

C) Assumptions of production region

The Production of the feedstock in each region is assumed (see Appendix 3). In Sweden the feedstock for ethanol production are wheat, barley, triticale and rye (Börjesson et al., 2013). Other EU countries supplies the Swedish ethanol production (SEA, 2011). In Brazil, ethanol from sugarcane is produced as well as from other unknown sugarcane producing countries in small amounts. In the other EU countries ethanol is produced from wheat, triticale, barley, rye, and sugar beets (SOU, 2013). From other unknown countries small amounts of that feedstock are assumed to be produced in EU. In the USA, ethanol is assumed to be produced from corn. The production in USA is assumed to have import of small amounts of corn for ethanol production from other countries. Assumptions are also made that other countries produces amounts of sugarcane, sugar beets and corn (see Appendix 3).

3.3.4 Fatty acid methyl ether production and feedstock analysis

Assumptions of FAME feedstock, the origin of the feedstock and production area are assumed according to the stepwise assumption in section 3.3.2.

A) Assumptions of feedstock

All FAME is produced from rapeseed feedstock (SEA, 2014a).

B) Assumptions of agriculture and feedstock origin

For the years 2011-2013 the rapeseed agriculture origin are given by sustainable fuels report(2014c). Besides from Australia, Europe is the dominating rapeseed producer in 2013. Only small amounts are cultivated in Sweden (see Appendix 5).

C) Assumptions of production region

Sweden has produced RME since 2007, mostly by Perstorp AB (SEA, 2014a). The production is dominated by EU and Sweden imports around half its consumption volumes from Lithuania, Denmark and other EU countries(SEA, 2014a). For FAME- assumptions see Appendix 5.

3.3.5 Hydro treated vegetable oils production and feedstock analysis

Assumptions for HVO feedstock, the origin of the feedstock and production area are assumed according to the stepwise assumption in section 3.3.2.

A) Assumptions of feedstock

Feedstock for HVO fuels are given by The Swedish Energy Agency (SEA, 2014c). The HVO feedstock is mostly waste from animal residues, palm oil and tall oil. Before 2011 there were no consumption of HVO (SEA, 2014d).

B) Assumptions of agriculture and feedstock origin

The origin of the HVO feedstock are given by the energy department’s sustainable fuel report (SEA, 2014c). For values of HVO feedstock see appendix 6.

C) Assumptions of production region

In Sweden Preem is producing the whole part of HVO, Preem produces HVO of tall oil (SEA, 2014a; Barr, 2010) The other amounts of HVO is produced in EU (SEA, 2014c)

3.3.6 Biogas production and feedstock

According to the energy department, 95% of all biogas was produced in Sweden by Swedish feedstock in 2013 (SEA, 2014c). The composition for biogas was 36% from sewage sludge, 30% food residues and 20% waste from slaughter house and agriculture waste, the rest was unknown but it was assumed to be agricultural residues (SEA, 2014c). The same biogas composition is used for all years in the accumulated LCA.

3.3.7 Fossil fuel production and feedstock analysis

Diesel and gasoline are produced from crude oil from different regions (SEA, 2014d). The origin of the crude oil has been collected through statistic Sweden database (SCB, 2015). The amounts of consumed diesel and gasoline were collected from the Swedish Energy Agency's report: Fuel use by the transportation sector (SEA, 2014d, 2008). The collected data is shown in Appendix 7.

3.3 Life cycle assessment

In this chapter the procedure for the accumulated life cycle assessment is described. A LCA method has been applied, see Section 2.3.

Goal and scope

The goal of the accumulated LCA is to provide the yearly impacts from Swedish biofuel consumption expressed in feedstock and production by region. The LCA will compare the impacts between fossil fuels and biofuels. The intended audience is researches within biofuels and policymakers. The purpose of the study is to investigate Sweden's impact according to the SEPA's consumption based patterns: used to portray trends and provide understanding of the biofuel impact both inside and outside the country. The questions answered by the accumulated LCA are:

- What are the impacts of the Swedish biofuel consumption between 2000 and 2013?
- Where do the impacts occur?

Those questions will provide answers to objective II) "To account for the impacts caused from biofuel consumption in Sweden from 2000-2013" and objective III) "To outline potential implications of limiting the impact categories".

To put the accumulated LCA in a larger context, another question is:

- What is the difference in impacts if fossil fuel should have been used instead of biofuels?

The assessed products are the conventional biofuels: ethanol, FAME, biogas and HVO. The fossil fuels consider diesel and gasoline. All the products are intended for transportation purposes, excluding fuel consumption in aircraft, ship and building machines. The functional unit is the total amount of consumed fuel per fuel type and year.

The comparison with fossil fuels is obtained through the same energy content as the functional unit.

The study is an attributional LCA and not consequential. E.g. meaning that average data is used and ILUC is not included.

System boundaries

The multiple LCAs were done with a cradle to gate perspective, equal the term "well to tank". This means that all impacts of the upstream process are included in the assessment: material extraction, production and distribution of the fuels in Sweden. Emissions from combustion are included in the comparison with fossil fuels, this is because the emissions of fuel combustion are considerable higher for combustion of fossil fuel than biofuels.

The system boundaries can be seen in the simplified flowchart in Figure 7. All boxes contain several sub steps and processes. The dotted line indicates the system boundary. The boundary covers the total fuel product phases. The border between nature and the technological system is crossed when a resource is coming from nature e.g. crude oil is extracted or when a substance leave the system and enters air, soil or water.

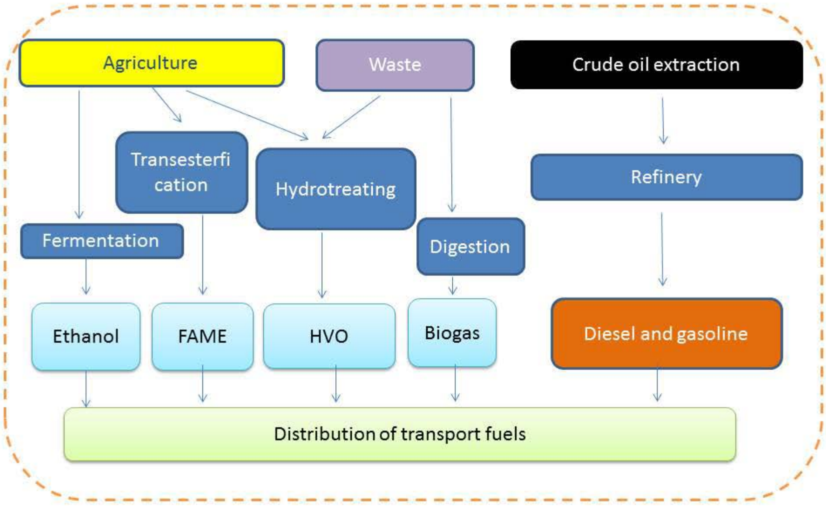

Figure 7: Flow chart of bio and fossil fuels

The geographical boundaries are not specific. A figure of the foreground system is shown in Figure 8 where the life cycle geographical pattern is showed. The time horizon is the past years between 2000 and 2013. The time horizon for the impact categories has a longer time horizon. E.g. global warming potential is calculated for a 100 year period.

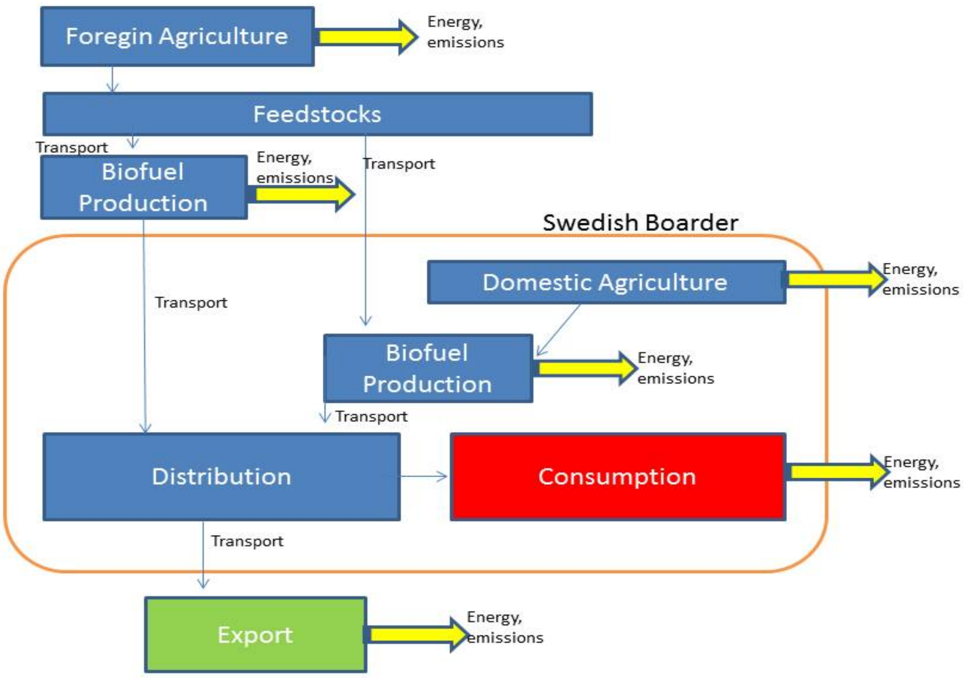

Figure 8: Geographical flow chart

Impact assessment indicators

The indicators chosen should give as a complete picture as possible of the impacts from the fuels. Many of the influential indicators on biofuels are involving local conditions and agriculture studying

(Laurent, Olsen and Hauschild, 2012; Wiloso, Heijungs and De Snoo, 2012). The agriculture is hard to get a comprehensive assessment on. Climate change, eutrophication, acidification, freshwater toxicity and photochemical oxidant formation are chosen since they cover the most significant impacts (Buchholz, Luzadis and Volk, 2009) and are considered important by EU's Joint Research Centre (JRC, 2011). Land use, land use change and indirect land use change is not included. Nevertheless, they are of importance, the reliability in the datasets is assumed to be weak for land use and the LCIs modelled from literature do not include land use.

3.4 Life cycle inventory

Even though there is a large amount of published LCAs, not many of them are transparent about their LCIs. This involves problem if they should be reused in other research. F3 fuels has a database made by Tivander et al. (2013) with LCI datasets for the most of the Swedish produced biofuels. The quality varies and only HVO from rapeseed and palm oil where sufficient. The other F3 fuel LCIs' had shortages since the energy sources were unclearly documented. All datasets were modified in GaBi software in order to share the production steps of feedstock and production. Transports to Sweden were also included. A brief overview of the production datasets used is shown in table 3.

Table 3 General overview of datasets

Dataset	Conditions	Used for	Allocation	Main source
Ethanol, wheat	SE	SE	Energy	(Martin, 2015)
Ethanol, Rye	EU	EU	none	Ecoinvent Database
Ethanol, sugarbeets	CH	EU, others	none	Ecoinvent Database
Ethanol, sugarcanes	BR	BR, others	None	Ecoinvent Database
Biogas, seawage sludge	CH	SE	None	Ecoinvent Database
Biogas, agricultural residues	CH	SE	None	Ecoinvent Database
Biogas, household waste	CH	SE	None	Ecoinvent Database
Rapeseed methyl ester	CH	EU, SE, others	Economic	Ecoinvent Database
HVO, from waste	FI	EU, others	None	(Reinhardt, 2006)
HVO, animal fat	FI	EU,others	None	(Reinhardt, 2006)
HVO, vegetable oils	EU, FI	EU, others	System exp.	(Tivander, 2013)
HVO, tall oil	SE	SE	Energy	(Barr, 2010)
HVO, palm oil	South Asia, FI	South Asia	System exp.	(Tivander, 2013)

3.4.1 Feedstock life cycle inventory

The datasets for ethanol are dependent on the feedstock, agriculture and production region. Feedstock is exported from some region for fermentation in other regions. Since there is no database providing processes for each country for the corresponding feedstock the datasets had to be simplified according to Table 4.

Table 4 Ethanol feedstock datasets

Ethanol agriculture datasets					
	SE	**EU**	**Brazil**	**USA**	**Other**
Wheat	Wheat-	Wheat grains-	-	-	Wheat grains-

	SE(Martin,2015)	EU(Ecoinvent)			EU(Ecoinvent)
Corn	-	-	-	Corn-US(Ecoinvent)	Corn-USA(Ecoinvent)
Sugarcane	-	-	Sugarcane-Br(Ecoinvent)	-	Sugarcane-BR(Ecoinvent)
Triticale	Wheat-SE(Martin et al.,2015)	Rye-EU(Ecoinvent)	-	-	Rye-EU(Ecoinvent)
Barley	Wheat-SE(Martin et al., 2015)	Barley grains-CH(Ecoinvent)	-	-	Barley grains-CH(Ecoinvent)
Sugarbeets	-	Sugarbeets-EU(Ecoinvent)	-	-	Sugarbeets-EU(Ecoinvent)
Rye	Wheat-SE(Martin et al., 2015)	Rye-EU(Ecoinvent)	-	-	Rye-EU(Ecoinvent)
Molasses	Wheat-SE(Martin et al., 2015)	Rye-EU(Ecoinvent)	-	-	Rye-EU(Ecoinvent)

The feedstock for RME is Rapeseed. The rapeseed agriculture is an aggregated CH dataset from Eco Invent called “EU rape oil at mill”. This dataset is applied on all rapeseed agriculture since there is no other rapeseed datasets available.

In Table 5 the feedstock for HVO are shown. The tall oil is a rest product from the pulp industry. It’s modified from an Eco Invent dataset called RER: Sulphate pulp ECF bleached. The burdens on tall oil should be mass allocated as 2% of the sulphate pulp according to (Venditti, 2012).

Table 5 HVO feedstock datasets

HVO feedstock			
	Region	Dataset	Allocation
HVO-tall oil	SE	Tall oil-SE	Mass
HVO-palmoil	South Asia	Malaysia Palm oil	System exp
HVO-rapeseed	EU	EU: Rapeseed agriculture	Economic allocation
HVO-animal fat	EU	CH:Tallow at plant	No allocation
HVO-hvo animal waste	EU	CH:Tallow at plant	No allocation

The datasets for Palm oil are based on system expansion and taken from an aggregated LCI made by Tivander (2013). HVO rapeseed feedstock are also from a LCI by Tivander (2013). The HVO from animal fat is approximated with an ecoInvent dataset for Tallow. This dataset is used as an approximation for animal fat. For HVO anima fat no feedstock is used, since the waste is supposed to be a rest product.

3.3.8 Production life cycle inventory

In Sweden the biofuels ethanol, RME, HVO and biogas are produced. The overall flow scheme shows the feedstock and production of biofuels in Sweden (see Figure 9).The Swedish ethanol production from grains is modelled from a LCA made of Martin et al. (2014) and modified in Gabi software.

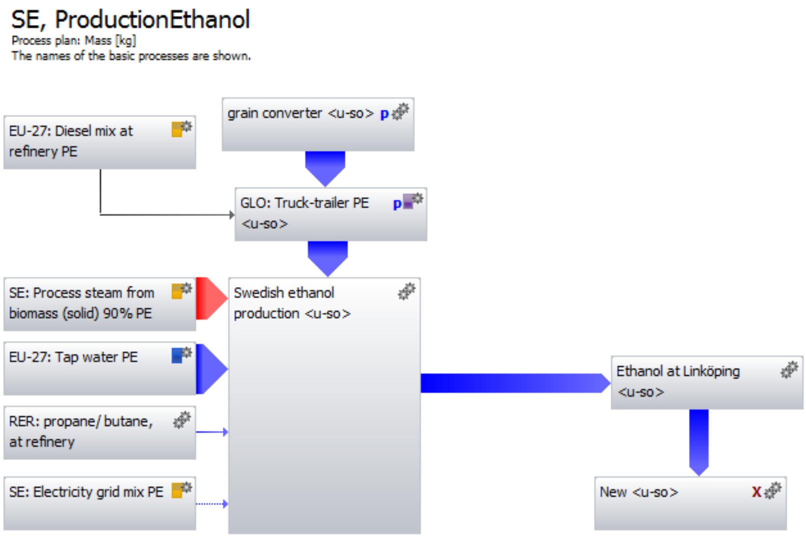

Figure 9: Ethanol production process in Sweden

The process "grain converter" includes both grains from Sweden, EU and other countries. The amounts of grains from each region are changing over the years according to appendix 3. One difference between this dataset (see Figure 9) and the LCA done by Martin et al. (2014) is that there is not any catalyst in this dataset. The outputs are besides ethanol, dried distillers grains, syrup, impurities, thin stillage and condensate. The allocation is based on energy allocation on these outputs (Martin et al., 2014).

The RME production is used both for Sweden and EU. The dataset for RME is applied from ecoinvent database, the dataset is named "RER: Rape methyl ester, at esterification plant". The inputs of rape oil changes yearly from different region see Appendix 7.

In Sweden all HVO produced is from tall oil (SEA, 2014c). There are no public reports of HVO tall oil production. The dataset had to be made from an energy and carbon dioxide report by Barr (2010). A process flow chart is presented in Figure 10. The feedstock of tall oil is calculated from an ecoinvent sulphate pulp process.

The raw tall oil is transported from the pulp and paper plant to Sunpine Fermentation plant in Skellefteå. In this process raw tall oil is transesterficated to raw tall diesel (FAME). The tall diesel is transported to Preem in Gothenburg for hydrofication, the end product is HVO (see Figure 10). The calculations for the inputs and outputs in the processes are presented in Appendix 8.

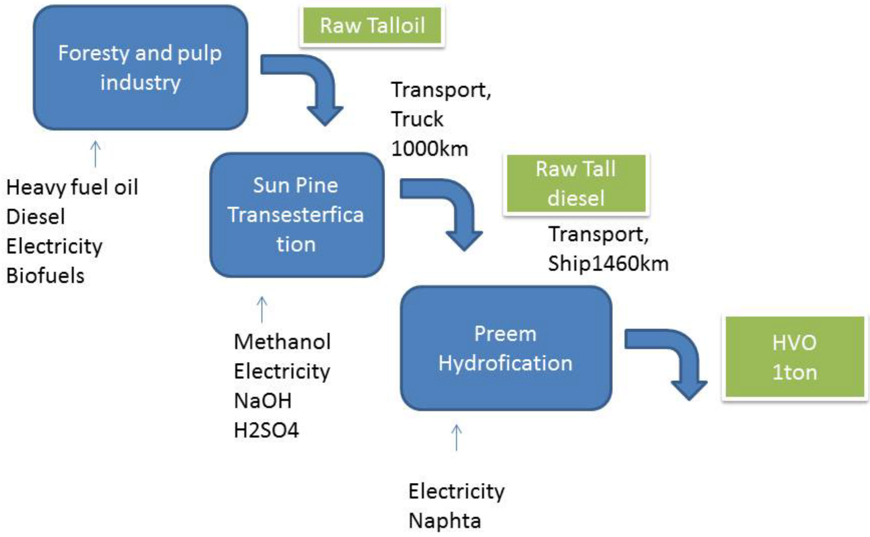

Figure 10: Production of HVO from tall oil

Biogas is produced in Sweden, ecoinvent datasets were applied (see Table 6).

Table 6 Datasets used for biogas production

Biogas Production mix				Source
CH: biogas, from agricultural co-digestion, not covered, at storage [fuels]	Standard volume	19	Nm3	Ecoinvent Database
CH: biogas, from biowaste, at storage [fuels]	Standard volume	20	Nm3	Ecoinvent Database
CH: biogas, from sewage sludge, at storage [fuels]	Standard volume	36	Nm3	Ecoinvent Database

The European production of ethanol from wheat, rye, triticale and barley are obtained from ecoinvent rye fermentation. This is a simplification since there are no datasets for barley, triticale and wheat ethanol production from EU in the accessible LCI-databases. The LCI-data for ethanol production from sugar beets was available in ecoinvent database.

For FAME production is the same dataset used as for Swedish European RME production. HVO are produced from palm oil, rapeseed oil, animal fat and animal waste. Production phases for palm oil and rapeseed oil could be found in F3 database (Tivander, 2013). For animal waste and fat rapeseed oil HVO production was modified from an energy and mass analysis of the “Povoo process” of NexBTL HVO production in Finland (Reinhardt, 2006). The modification and use of a simplified mass and energy analysis as LCI-dataset is a generalization, the production process is made for vegetable oils and not preliminary for animal residues, but no better LCI could be found. In figure 11 is the transesterification and hydrotreating by NExBTL shown. Instead of vegetable oil, animal fat was the input. The complete LCI for production of animal fat and waste can be seen in appendix 10.

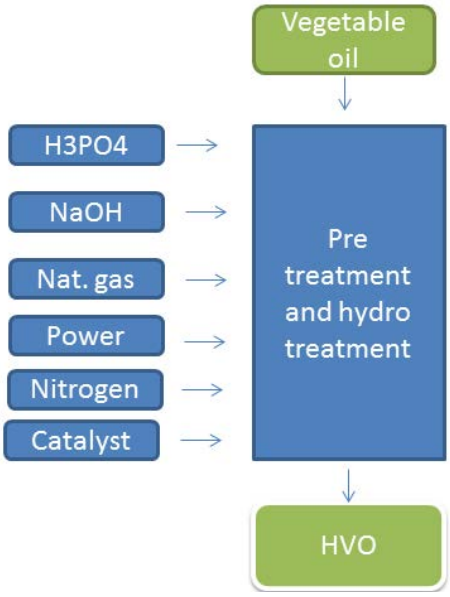

Figure 11: HVO production process

Only corn is assumed to be grown for ethanol production in USA. The used dataset is from ecoinvent 2.2 called "US: Ethanol from corn". The same dataset is used for "others corn". Brazilian production life cycle inventory is only based on the pre-made dataset "BR: Ethanol from sugarcane", with no allocation. Since sugarcane also is produced in other countries, it's assumed to be imported and produced in Brazi. In other unknown countries, ethanol from sugarcane, corn and sugarbeets are produced. All those datasets have been used in production in other regions.

3.4.3 Fossil fuels life cycle inventory

Datasets for fossil fuels are taken from PE-international: EU diesel at storage and EU gasoline. These datasets are not fully correct according to this assessment since they contain the associated resources and impact by its biofuel blend, small amounts of ethanol and FAME.

3.4.3 Combustion of the fuels

The combustion is a brief comparison to see the consequences of biofuel and fossil fuel combustion in Sweden. The impact categories considered are global warming, eutrophication, acidification and photochemical oxidant formation. Aquatic toxicity is not included since values could not be found. The environmental effects from the combustion are based on Börjesson et al. (2010). Simplifications are made so that all ethanol, biogas and gasoline are supposed to be combusted in light vehicles, RME and diesel in heavy vehicles. Since there are no data of HVO, the emissions of RME are used in the calculations.

4 Impact categories in Swedish environmental system analyses

The scope of the Swedish environmental system analyses on biofuels is interpreted in the following sections. The analyzed reports can be seen in appendix 1. The results are presented and followed by an analysis.

4.1 Results meta-study

From the 68 investigated ESA reports the assessed impact categories can be seen in Figure 12. Of the 68 articles 61 were on GWP, 50 energy or efficiency calculations, 19 EP and 17 AP. Land use change in any form is investigated in 18 cases. The TP is only considered in 3 reports and water depletion in one. The majority of the ESAs are assessing the GWP and energy efficiency ratios for production. Energy analysis is not an environmental impact category but is included since it's a frequently assessed indicator.

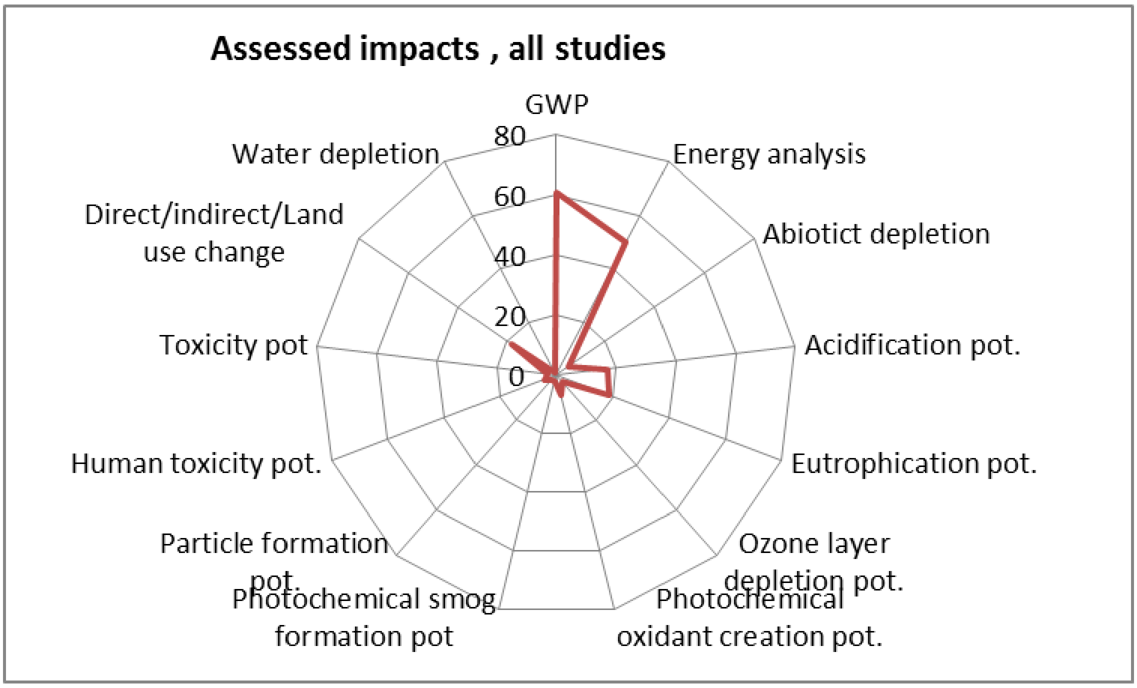

Figure 12: Assessed environmental impact categories in all reports

Out of the 68 ESAs, 34 were stated as LCAs. The assessed environmental impact categories in those reports can be seen in Figure 13. The relative scope of the LCAs is wider than for all reports (compare Figure 12 and 13). Again, the emphasis is dominating by GWP and energy efficiency. GWP as only impact category has been assessed in 13 of the LCAs. EP is included in 18 of the articles and AP in 16.

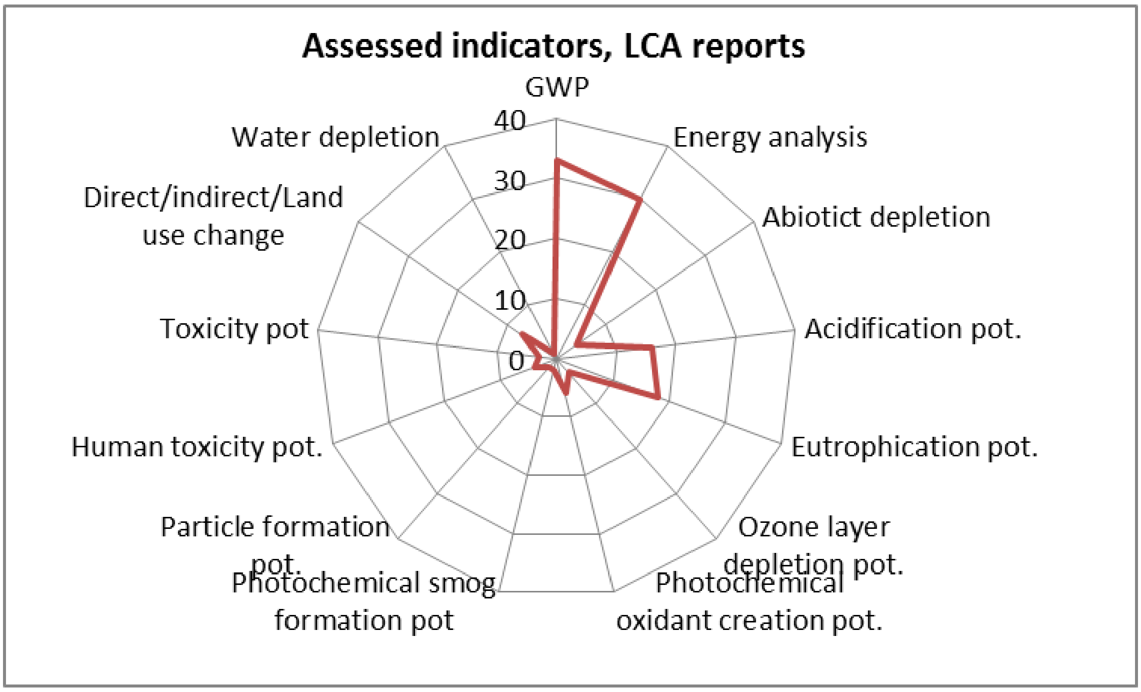

Figure 13: assessed environmental impact categories in LCAs

The distribution of studies assessing land use or land use change impacts can be seen in figure 14. Of 20 studies considering land use change 16 were calculating potential emissions from the arable land transformed.

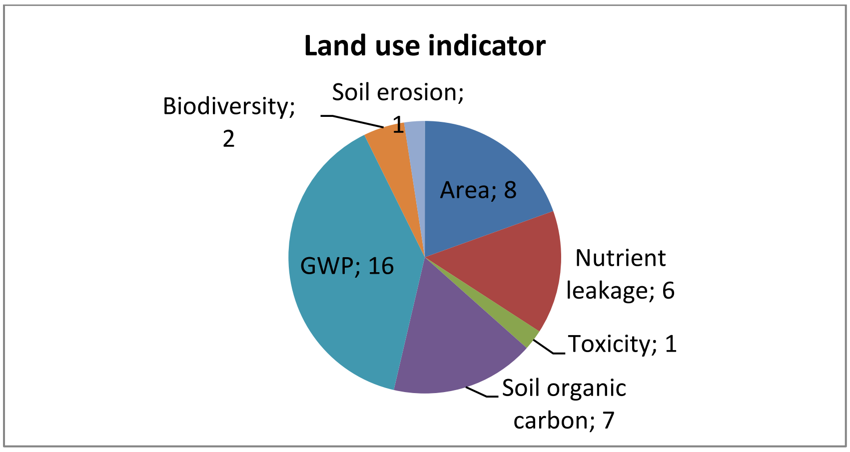

Figure 14: Land use environmental categories in all studies

The included environmental impact categories included in the conclusions of the reports can be seen in Figure 11. GWP is included in 57 reports, energy efficiency in 26, AP in 10, EP in 10, Land use or land use change in 15.

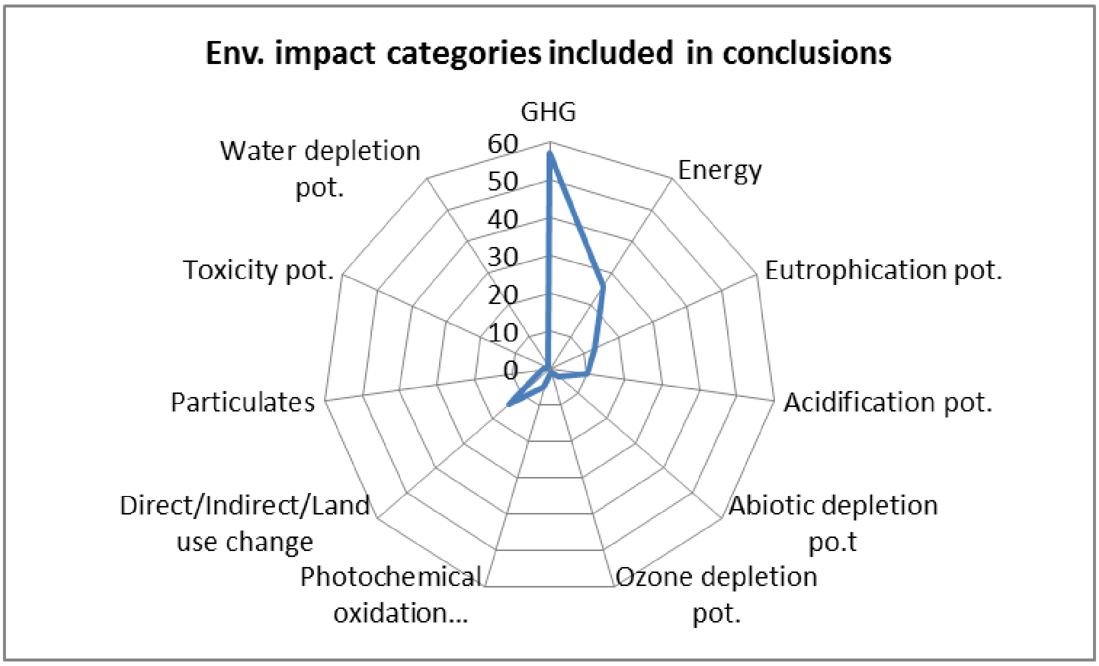

Figure 15 Included environmental impact categories in conclusions

4.2 Analysis of META study

The analysis covers the assessed biofuels, a qualitative analysis for justification and discussed limitations in the studies. The limitations of this study will also be described.

As seen in the meta-study results, many of the LCAs and ESAs emphasize mainly on GWP. Only 23 of the studies are investigating at least one more impact category than GWP and Energy (see Appendix 1). Several of the research papers are both investigating several impact categories and base their conclusion on a more inclusive impact view (see Appendix 1).

4.2.1 Assessed biofuels

The numbers of assessments for each biofuel type can be seen in Figure 16. The most frequent assessed biofuel is ethanol. Biogas is the second most frequently assessed. Biodiesel production[2] studies were also apparent in many studies.

[2] Biodiesel includes FAME, DME, Fischer Tropsch and HVO.

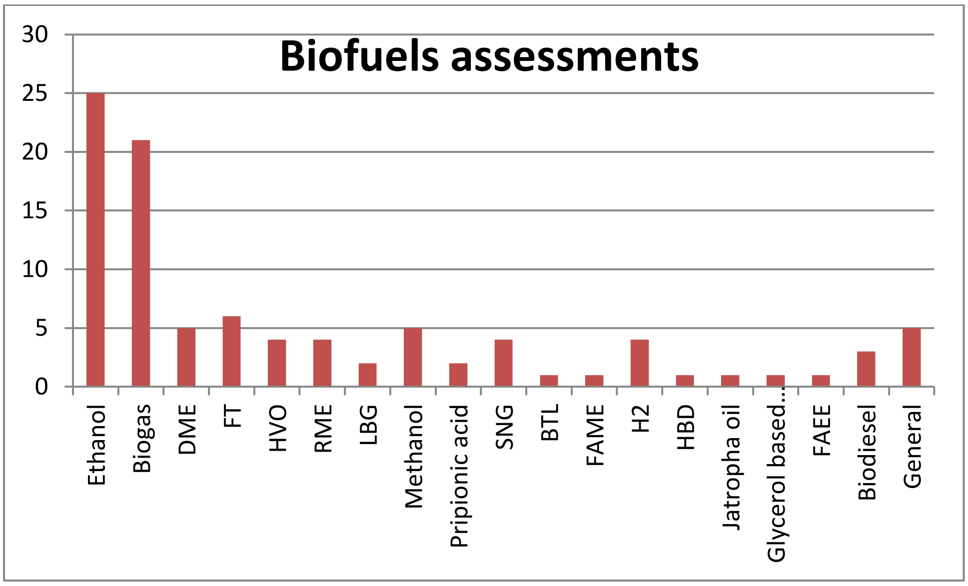

Figure 16: Biofuels assessed in the studies

4.2.2 Type of study

There were different methods applied on the assessments. Of all the studies 33 were stated as LCAs. In Figure 17 the type of study is presented.

A distinction shall be made between the articles that are based on LCA and the studies made to evaluate environmental impacts. The reports that did not have a LCA method were system analysis, scenario studies, reviews, land use, CO2 mitigation, critical assessment and recommendation, eco toxicity assessment and environmental assessment. The environmental assessments differ in scope from GWP and energy analysis to impacts on soil characteristics, water quality and biodiversity effects.

Scenario studies are the articles investigating potential pathways for biofuels under different conditions as a large scale bio based energy system.

The reviews discussed implications with assessments of GHG-emissions, land use calculations, indirect land use changes or summarized the state of art of a technology. The articles on critical assessment are showing drawbacks and gives recommendations.

The critical assessments have different scope and purposes e.g. the need to consider right type of land for crop agriculture to avoid emissions and land degradation effects, methodological choices to calculate soil emissions properly and LCA methods to cover more aspects to reach higher credibility of results, conflicts with food producing arable land and economic consequences.

One report is aiming at the Eco toxicity effects from pesticide use in biofuel feedstock production and is therefore classified as Eco toxicity assessment.

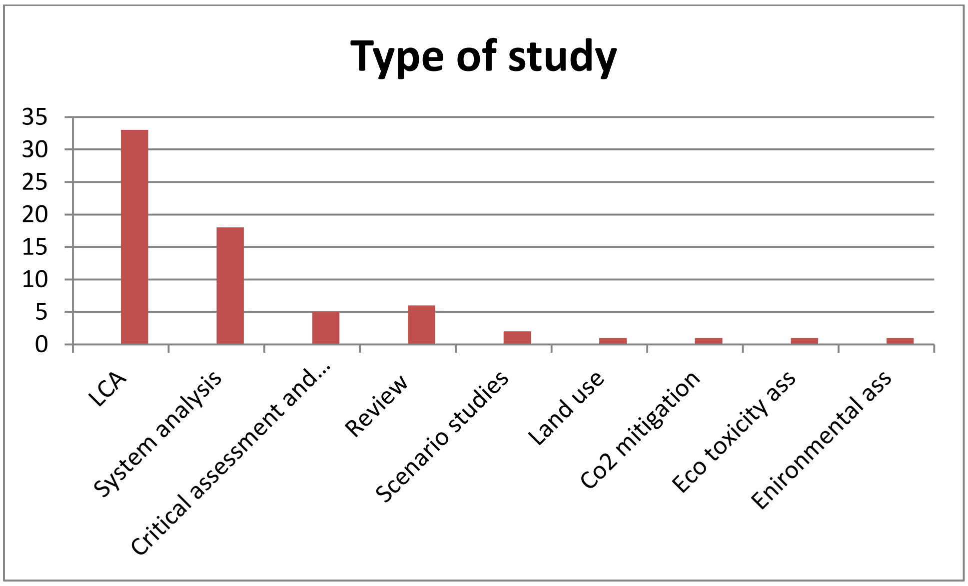

Figure 17: Type of study

4.2.3 Justification of impact categories in life cycle assessments

The LCAs have a standard to be followed. This standard claims that the selection of impact categories should be with motivation. The LCIA-method should also be stated. An overview of the LCAs LCIA method, environmental focus, discussion of limitations and discussion of other impacts can be seen in table 7. Among the LCAs there were 6 studies giving motivation for selection of impact categories. The lack of justification indicates that the ISO 14040 standard is not followed and that few of the LCAs on biofuels motivate their choices.

Bernstad and la Cour Jansen (2011) assesses GWP, AP, nutrient enrichment, OD, and POCP on biogas from household waste. The justification for inclusion of those impact categories was "These categories were chosen as they are environmentally relevant and internationally accepted with ISO 14042" (Bernstad and la Cour Jansen, 2011). Bernstad and la Cour Jansen's other report from 2012 assessed GWP, AP and EP. The authors Bernstad and la Cour Jansen refer to the Swedish environmental objectives SEPA (2007) and explains that those impact indicators are believed to be of large importance.

Lutherbacher et al. (2009) justifies the inclusion of GWP only since it's a parameter of interest in biofuel production without reference.

Hagman et al. (2013) explains that an important reason for using biofuels is their reduction of GHG. Water use is also assessed by Hagman et al. (2013) motivated by worries for scarcity of water according to (Berndes, 2002).

The study performed by Arvidsson et al. (2011) is a comparison LCA for different feedstock for HVO production. The choice of impact categories is justified from a literature study of other LCAs on ethanol compared to fossil fuels. The conclusion from the literature review was that many additional impact categories should be included besides GHG emissons in LCAs on biofuels. Eventually, GWP (including soil emissons), AP and EP was assessed (Arvidsson et al., 2011).

Further examples of writers applying the recommendations by ISO 14047 are Börjesson and Tufvesson (2011). A broad literature review gives the state of the art. Eutrophication is highlighted in correspondence to the plantary boundraries defined by Rockström et al. (2009), and the authors argue for the need to include eutrophication in assessments where land use change is occurring. There is no motivation for assessing GWP.

Table 7 Motivation, limitations and discussion of other impacts in LCAs

Author	Year	LCIA-method	Motivation for selection of impact categories?	Discussing limitations of impacts?	Discussing other impacts?
Bernstad et Al.	2011	n.d	yes	Yes, toxicology e	yes
Lutherbacher et	2009	Eco Indicator 9	yes	no	no
Bernstad et Al.	2012	EDIP 2003	yes	Yes, that the ass	no
Hagman et al.	2013	n.d	yes	LUC,AP,EP,wate	yes
Arvidsson et al.	2010	n.d	yes	no	no
Börjesson et al.	2011	n.d	yes	Yes, biodiversity	yes
Bengtsson et al.	2012	n.d	n.d	no	no
Ahlgren et al.	2008	IPCC	n.d	no	no
Ahlgren et al.	2009	IPCC	n.d	no	yes
Bernesson et al.	2004	n.d	n.d	no	no
Brynolf et al.	2014	n.d	n.d	Yes, NOX, LUC	yes
Ekman et al.	2011	n.d	n.d	Yes uncertaintes	no
Ekman et al.	2013	n.d	n.d	no	yes
Garraín et al.	2014	EPD 2008	n.d	yes, iLUC.defore	yes
Gonzalez Garzia	2011	CML 2 baseline	n.d	no	yes
Karlsson et al.	2014	n.d	n.d	nitrate, phosour	yes
Khatiwada et al.	2011	IPCC	n.d	Soil carbon	no
Kimming et al.	2011	IPCC	n.d	Soil carbon	no
Martin et al.	2014	EPD 2008	n.d	no	no
Tidåker et al.	2011	IPCC	n.d	Soil carbon, ILUC	yes
Tufvesson et al.	2013	IPCC 2006	n.d	yes; No iLUC	yes
Wang et al.	2013	n.d	n.d	no	no
Lantz et al.	2013	n.d	n.d	no	no
Larsolle & Ande	2013	IPCC (2007)	n.d	no	no
Bernesson et al.	2006	IPCC	n.d	no	no
Tufvesson et al.	2013	n.d	n.d	no	no
Karlsson & Börje	2013	n.d	n.d	no	yes
Gonzales Garcia	2012	CML 2 baseline	n-d	no	no
Janssen et al.	2014	CML	n.d	no	no
Joelsson and Gu	2012	n.d	n.d	no	no
Bezergianni et a	2014	n.d	n.d	no	no
Suer et al.	2011	Recipe endpoi	n.d	yes; helth risks,	yes

Justification of impact categories in studies non-Life Cycle Assessments

Out of the ESAs that were not described as LCAs few gave motivation for theirs' selection of impact categories. Only 7 out of 29 studies gave any kind of motivation (see Table 8).

Two studies mention gap in research for inclusion of their impact categories. Those are Nordborg et al. (2014) and Risén et al. (2013) A toxicity assessment is performed by Nordeborg et al. (2014) to show on the impacts of pesticide use in biofuels. Nordeborg is criticizing the pre made LCI datasets for not being accurate. Risén et al. 2013 is motivating the choices to also include nutrients, besides of GHG and energy since it's not done before in assessments of biogas from algae.

A scenario analysis was performed by Börjesson et al. (2013). The aim was to assess different future scenarios for large scale biofuel. The scope to include only GWP in the assessment is motivated by the national and international targets on GWP performance.

Sparovek et al. (2008) is clearly motivating the need for assessments of LUC and ILUC connected to biofuels in Brazil. A problem definition is presented and motivated from previous research. The study is not accounting the possible emissions since the aim is to account for the area in terms of LUC and ILUC.

Hansen et al. (2013) is giving an comprehensive overview of LUC and biofuels. The impact areas accounted for are SOC (soil organic carbon), biodiversity and nutrient leakage. The authors comment that many studies of LUC are only considering GHG. Hansen et al. (2013) further remarks that there are many important environmental and social aspects to consider besides GHG when assessing LUC in connection to biofuels.

One assessment, made by Englund et al. (2012) is performed to fulfill obligations from the RED criteria 2009/28/EC (European Parliament, 2015) on short rotation coppice. The RED criteria requires values for GHG emissions and obligations to consider carbon stock and biodiversity impacts.

Table 8: Motivation, limitations and discussion of other impacts in non-LCAs

Author	Year	Environmental impacts chosen beacuse?	Disscussing limitations of impacts?	Discussing other impacts?
Börjesson et al.	2013	CO2 target	no	no
Börjesson and Mattiass	2008	n.d	yes, methane, nitrogen	yes
Börjesson et al.	2009	n.d	Yes, biodiversity, LUC	yes
Sparovek et al.	2008	Conserns of land use ch	no	no
Nordborg et al.	2014	gap in research	yes; human tox, ghg, ac	no
Risen et al.	2013	gap in research	no	yes
Samiei and Fröling	2014	gap in research	no	no
Hansen et al.	2013	land use affects	no	yes
Englund et al.	2012	RED-directive	no	yes
Gustavsson & Karlsson	2006	n.d	no	no
Johansson et al.	2014	n.d	no	no
Khatiwada et al.	2012	n.d	Yes; soil carbon, ILUC	yes
Caspeta et al.	2013	n.d	no	no
Wang et al.	2012	n.d	no	yes
Börjesson et al.	2013	n.d	yes, the indirect effects	yes
Alfors et al.	2010	n.d	no, outside the scope	no
Lubbe et al.	2012	n.d	no	no
Andersson & Harvey	2006	n.d	no	no
Naqvi et al.	2013	n.d	no	no
Pettersson and Harvey	2012	n.d	Yes; SOC	no
Bauer & Hultenberg	2013	n.d	no	no
Brau et al.	2013	n.d	no	no
Wetterlund et al.	2012	n.d	no	no
Börjesson et al.	2013	n.d	Yes; biodiversity, soil d	yes
Moghaddam et al.	2013	n.d	no	yes
Margeot et al.	2009	n.d	yes	yes
Björklund et al.	2001	n.d	Yes, toxicology, acids et	no
Ahlgren & Lucia	2014	n.d	no	no
Lageveld et al.	2012	n.d	yes, futher investigatio	yes

4.2.3 Analysis of limitations in the studies

Some of the investigators state their limited choice of impacts, whereas other don't See Table 7 and 8. Many of the studies claim that they are assessing environmental impacts but only consider GWP with no explanation about this choice. Any kind of environmental limitation is mentioned in 20 of the reviewed studies (see Table 7 and 8). Of those studies soil organic carbon (SOC) is declared as limitation in 6 studies, LUC and ILUC in 9, toxicity in 5, biodiversity in 3. Other limitations considered water depletion, nutrient leachate and other impacts.

LUC and ILUC can cover aspects of many impacts. Börjesson et al. (2009) only assesses GHG performance but discuss the need for supplementary assessments on biodiversity and social aspects on bioethanol due to LUC. Khatiwada et al. (2012) states that ILUC should be accounted for in GHG assessments. The reason for exclusion was methodological divergence in research. Other studies are just stating that their assessments exempted GHG emissions from LUC and ILUC e.g. (Hagman et al., 2013; Brynolf el al., 2014). One study assessing biogas production from residues states that their utilization is not causing ILUC (Tufvesson et al, 2013). Tidåker et al. (2014) is including LUC in the assessment but not ILUC. ILUC is exempted since there is not any calculation acceptance.

4.2.4 Limitations of the meta-review

All included articles are affecting the result in Figure 13. The limitation of the study is the aggregation with studies with different aims and methods. On the other hand, all studies are assessing environmental impacts in correlation to biofuel production. The system boundaries are different and may affect the outcome of the meta-study. E.g. for biogas production that can have a feedstock of household waste, whose assessments has narrow borders and doesn't include cradle emissions. The analyses with aim to compare production methods could also be limited in life time scope to exclude the agriculture phase.

5 Fuel consumption figures

For all years, Table 9 shows the total amount of consumed biofuels for transport purposes. The years from 2000 to 2006 are gathered from the publication: Fuel use for the transport sector (SEA, 2008). For the years 2007-2010 from: Fuel use by the transport sector (SEA, 2014d). The consumption volumes from 2011-2013 are from the sustainable fuels report (SEA, 2014c, 2013b, 2012). It should be highlighted that the values from the fuel use by the energy department differs 2011-2013 compared to their sustainable fuel report, the values from Sustainable fuels report is used. The energy department does not have any answer for this data error.

Table 9: Consumption of transport fuels in Sweden (SEA, 2014d) **and** (SEA, 2014c)

Fuel	unit	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
Ethanol	1000m3	26	42	76	150	261	285	321	358	422	389	400	420	407	355
FAME	1000m3	n.d	n.d	5	5	9	11	65	129	160	194	207	224	252	240
Biogas	Mm3	5	6	9	11	13	16	24	28	34	42	59	75	83	90
HVO	1000m3	0	0	0	0	0	0	0	0	0	0	0	49	131	290
Gasoline	1000m3	5335	5357	5405	5369	5204	5162	5063	4993	4604	4520	4237	3915	3593	3536
Diesel	1000m3	2529	2550	2921	3058	3385	3582	3662	3777	3860	3762	4123	4217	4092	4595

The amounts presented in GWh over the years can be seen in Figure 18. The biggest change over the years is that gasoline has decreased and diesel increased. The biofuels shows an increase over every year. Ethanol is showing the largest decrease and HVO has rapidly increased since 2011.

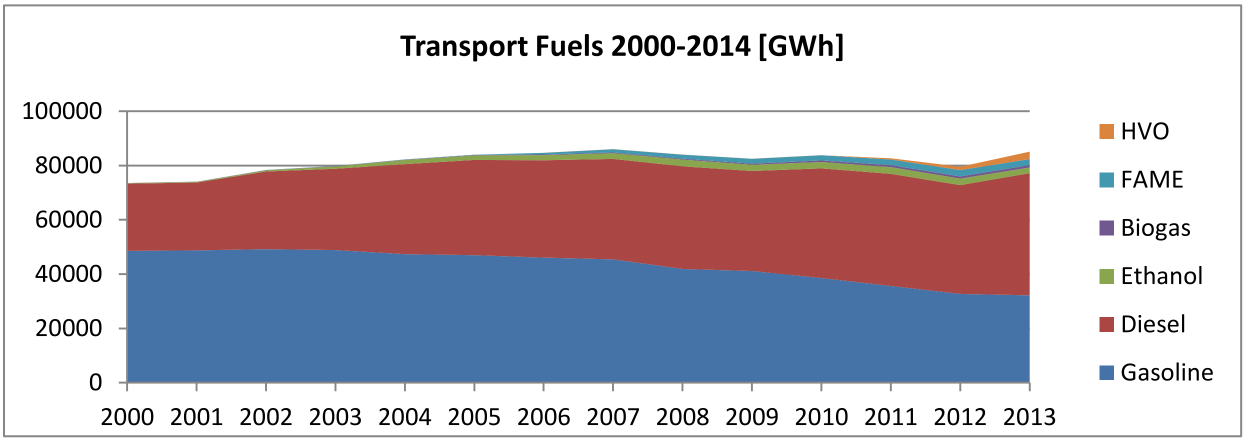

Figure 18 Consumed transportation fuels in Sweden 2000-2013

Figure 19 portrays the biofuel consumption in GWh. As mentioned, only ethanol has decreased whereas all other fuels show steady increase. Especially HVO consumption shows a large increase the last years.

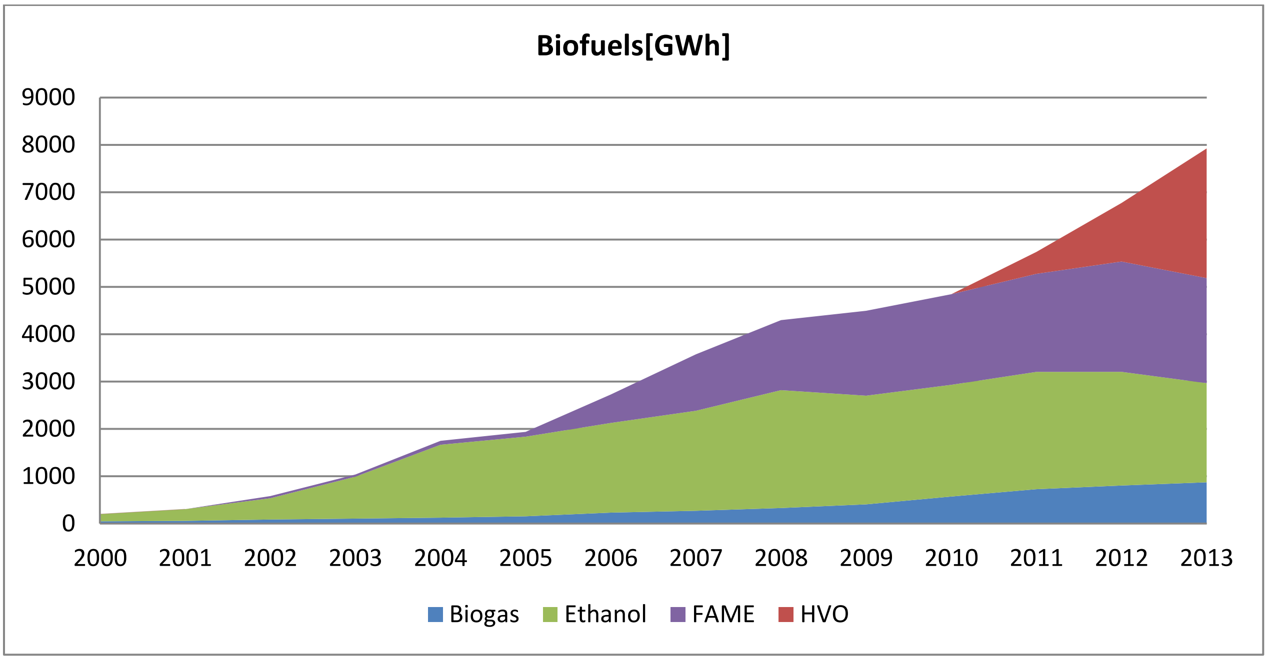

Figure 19 Energy content in consumed biofuels

Feedstock origin for all biofuels

When all the feedstock and their origin were assumed, the feedstock per region can be studied (see Figure 20).This Figure gives the feedstock origin over the years. EU and other countries are increasing their export to Sweden, there is also more domestic agriculture and production. This domestic agriculture is mainly from tall oil and wheat production. South Asia has been exporting palm oil for HVO production from 2012. From other countries the import consists of corn, sugarcane, sugar beets and rape seed.

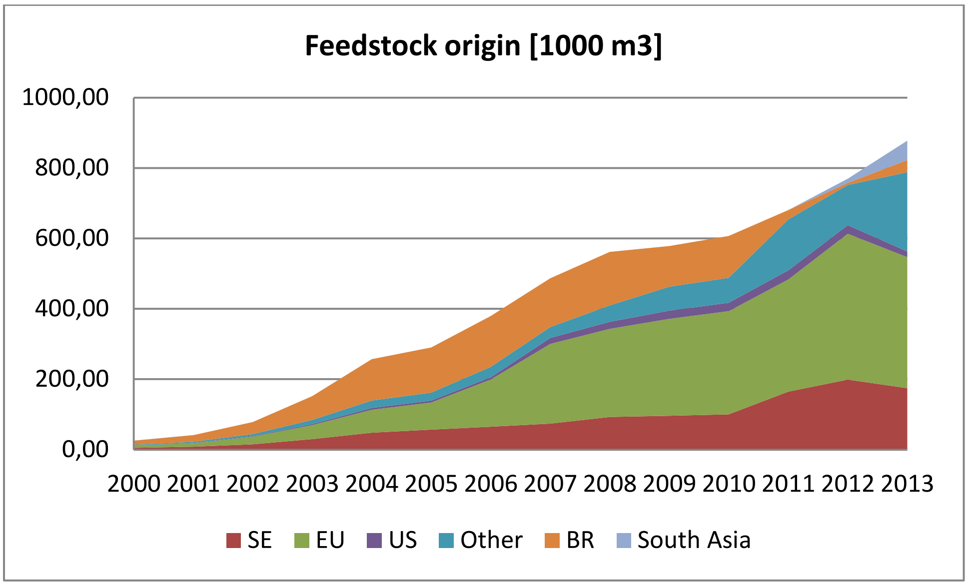

Figure 20: Feedstock origin for biofuels

In Figure 21 the importation origin of crude oil can be seen. Russia has been the dominating export country for crude oil to Sweden the last five years, Norway is another big exporter.

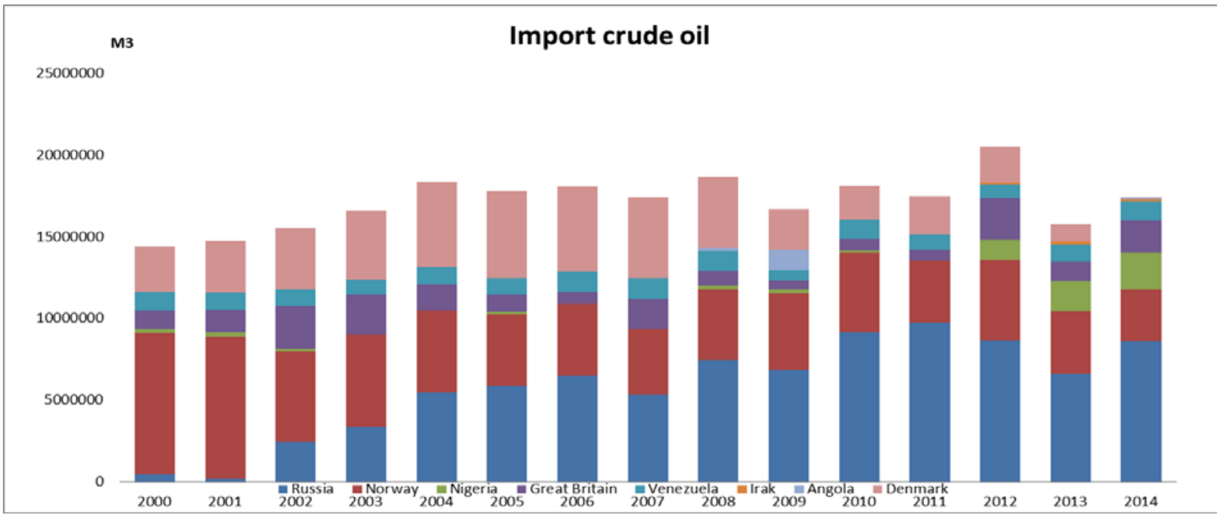

Figure 21 Import volume of crude oil per country

5 Assessed impacts for Sweden's biofuel consumption

In this section the accumulated environmental impacts for GWP, AP, EP, freshwater toxicity and POCP are presented as feedstock production (agriculture) and production of the fuels. The environmental impact potentials are presented as well to gate. The comparison with fossil fuels scenario is for the whole life cycle.

An analysis of the results is presented in section 5.6. The analysis presents the impact per MJ of fuel and impact per kg of feedstock. The analysis includes a comparison with other studies.

5.1 Global warming emissions for the total consumption

Figure 22 shows that the accumulated GWP follows the feedstock pattern (see Figure 20), with some exceptions: Brazilian contribution is low on GWP emissions and Sweden's domestic agriculture gives low values on GWP compared to its feedstock production part. The reason why Swedish production is large is due to the production of biogas. The fluctuations are larger the years of 2011-2013 when origin amounts of total feedstock per region were based on the Energy Agency (SEA, 2014c). During these years, the EU and Swedish agriculture amount is decreasing and is mostly replaced with HVO with lower GWP impacts. The trend of the latest years is to import more rapeseed from Australia and palm oil from Malaysia and Indonesia, which are both included in the figure's area of other countries agriculture.

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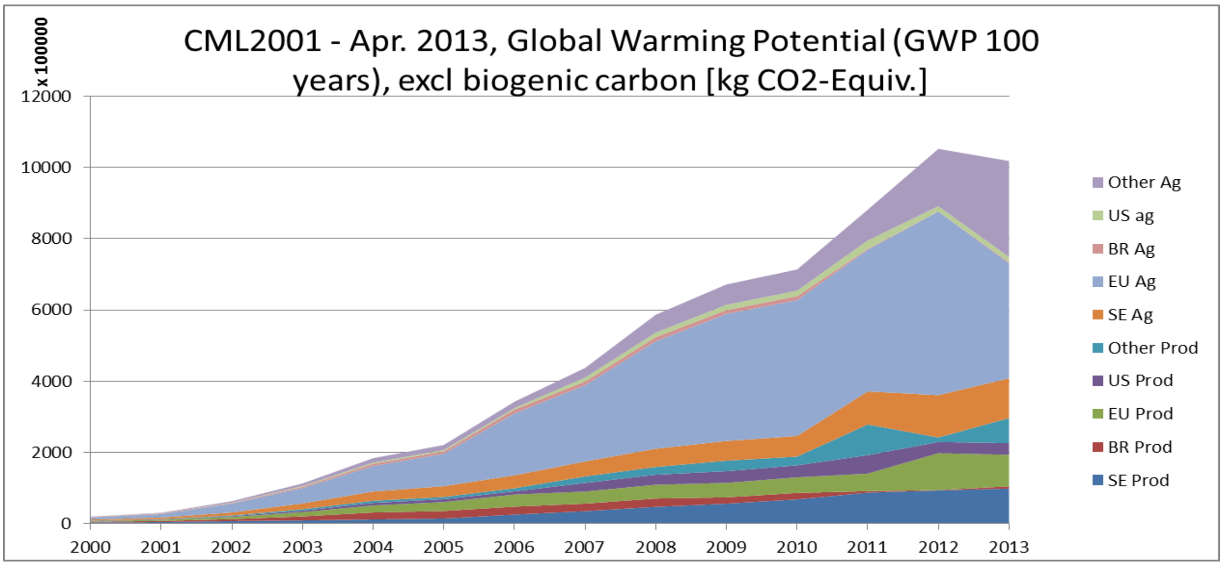


Figure 22: Accumulated GWP

In comparison to fossil fuel based transport system the biofuels emit fewer GWP emissions. Figure 23 shows the whole life cycle from cradle to the combustion of the fuels. The accumulated saving of GWP is more than two times compared to biofuels. The change for 2012 and 2013 are due to increased HVO from waste with a lower GWP.

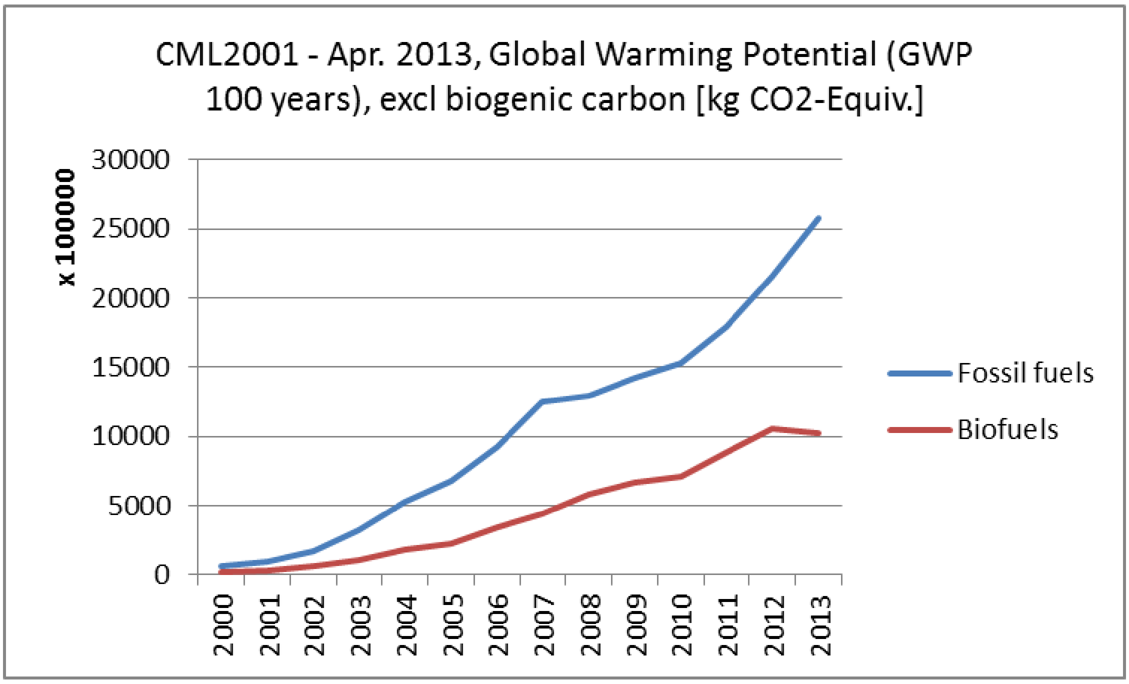


Figure 22 Global warming potential compared to fossil fuels

5.2 Acidification emissions for the total biofuel consumption

Figure 24 shows the yearly assessed influence of AP-eqv. for all biofuels. The EU has been recipient for the most of the AP emissions since the start of biofuel consumption. From 2007 and forward there has been an increase in the import of feedstock and biofuels from other countries (see figure 20). The feedstock from other countries of corn, rapeseed and sugarcane gives high impacts on the AP (see Section 5.6.3). It's worth noticing the small contribution of AP from the Brazilian agriculture, the import of sugarcane has been high but not the acidification emissions. Of the production step, Sweden gives the largest contribution.

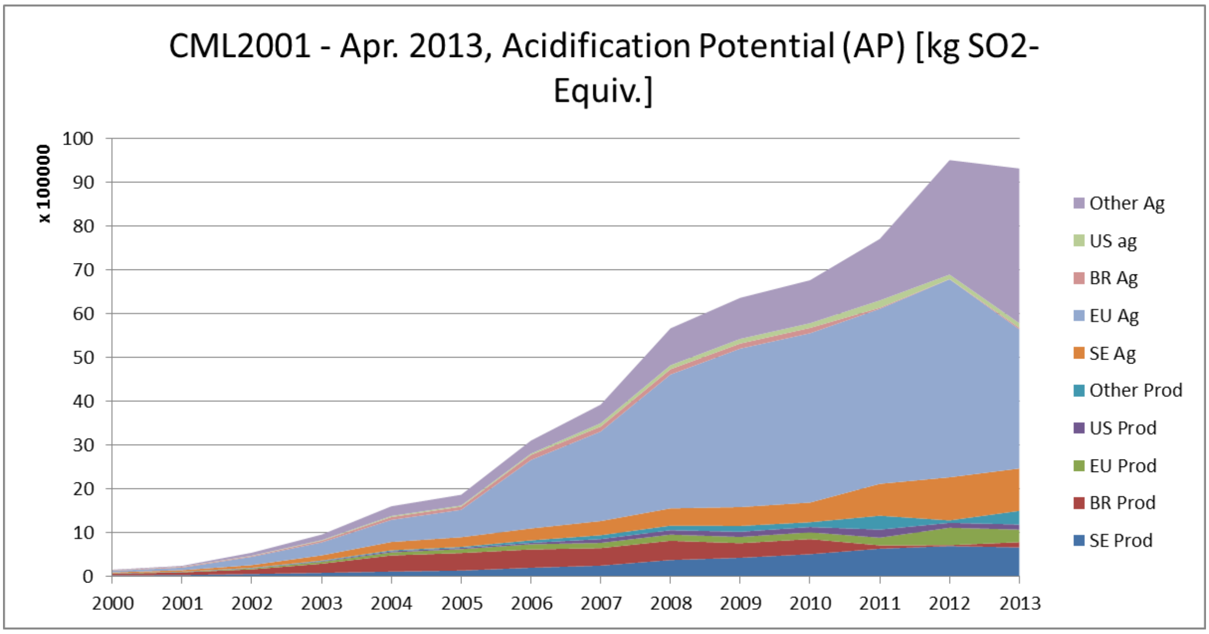


Figure 24: Yearly aggregated acidification potential

Compared with the complete lifecycle, the well to wheels AP is much higher for the biofuels (see Figure 25). The AP was almost the same as for the fossil fuel scenario but in 2005 larger part of ethanol from EU grains and Rape seed for RME was consumed, leading to larger AP emissions.

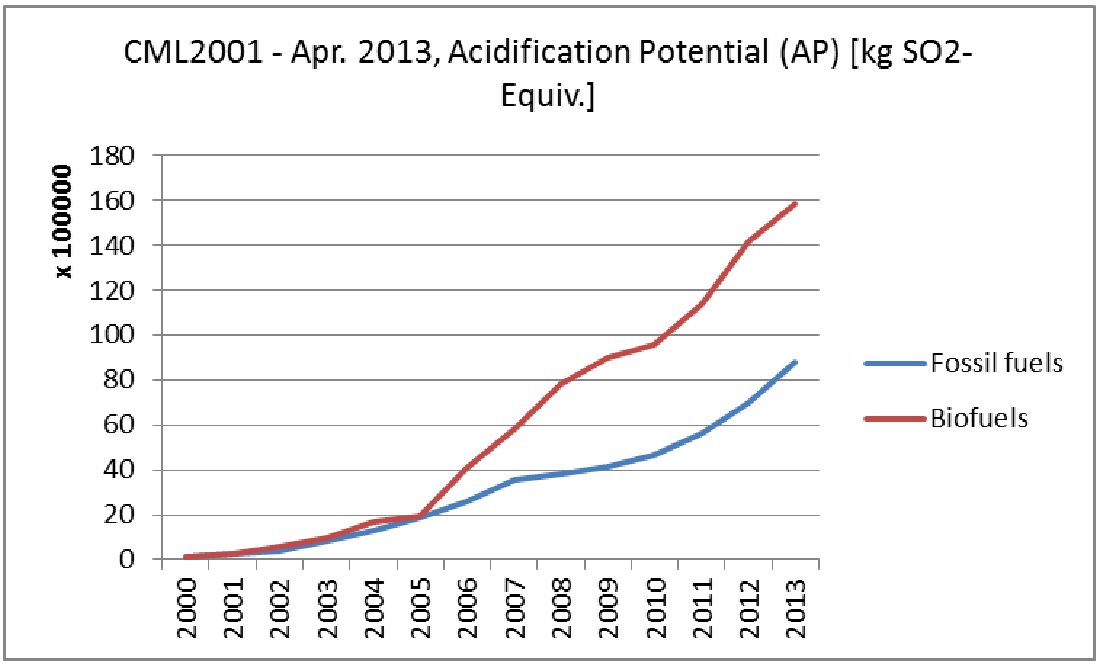


Figure 25: Acidification potential comparison between bio and fossil fuels

5.3 Eutrophication emissions for the total biofuel consumption

Figure 26 shows the EP emissions from 2000 to 2013.The assessed impact of eutrophication potential is largest in EU for all years. In the EU, grains for ethanol and rapeseed for RME and HVO production give high values on EP. Both EU grains and rapeseed are imported in large scale and is the central reason for EU's agricultural EP contribution. From other countries both the volumes of palm and rapeseed oil are large and gives high EP effects. The effect of Swedish agriculture is lesser as the datasets for Swedish ethanol grains and tall oil is low on EP influence. Brazilian ethanol gives small EP impact, even though large amounts of sugarcane ethanol are imported (see Section 5.6.5).

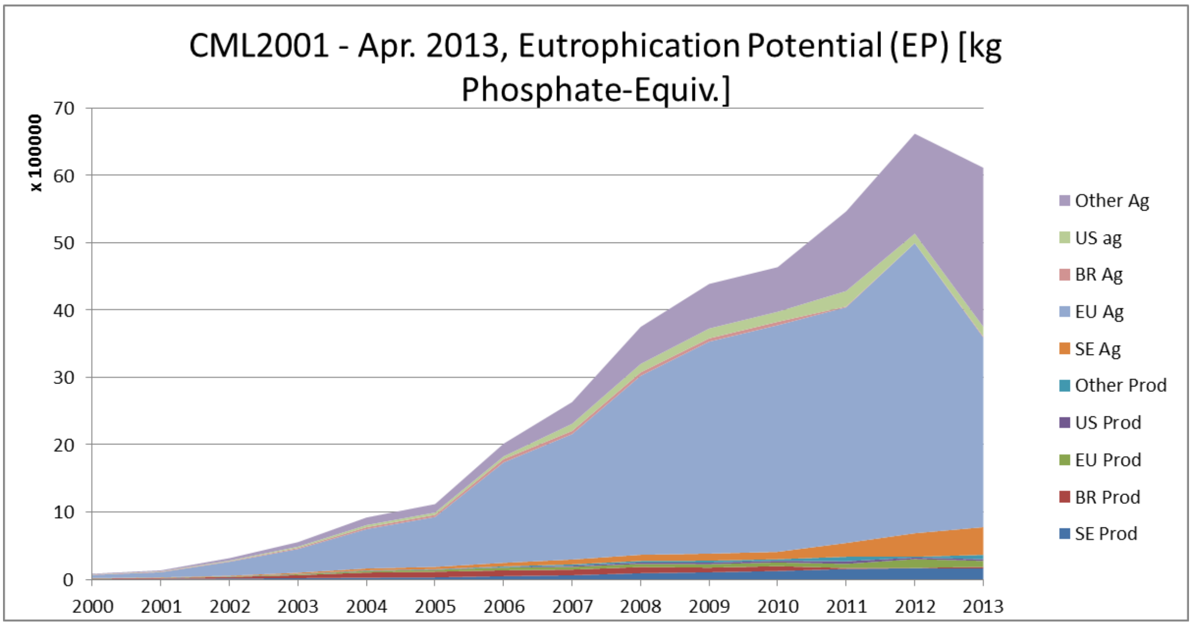


Figure 26 yearly EP for biofuels.

The eutrophication potential has been higher all years for the biofuels compared to the fossil fuel scenario. The only decrease has been the last years as result of the increase in HVO fuels. All HVO fuels besides of HVO from rapeseed give low EP emissions. Accumulated, the AP for biofuels is 5.6 times that of fossil fuels. The EP difference is the largest assessed impact indicator between biofuels and fossil fuels (see Figure 27).

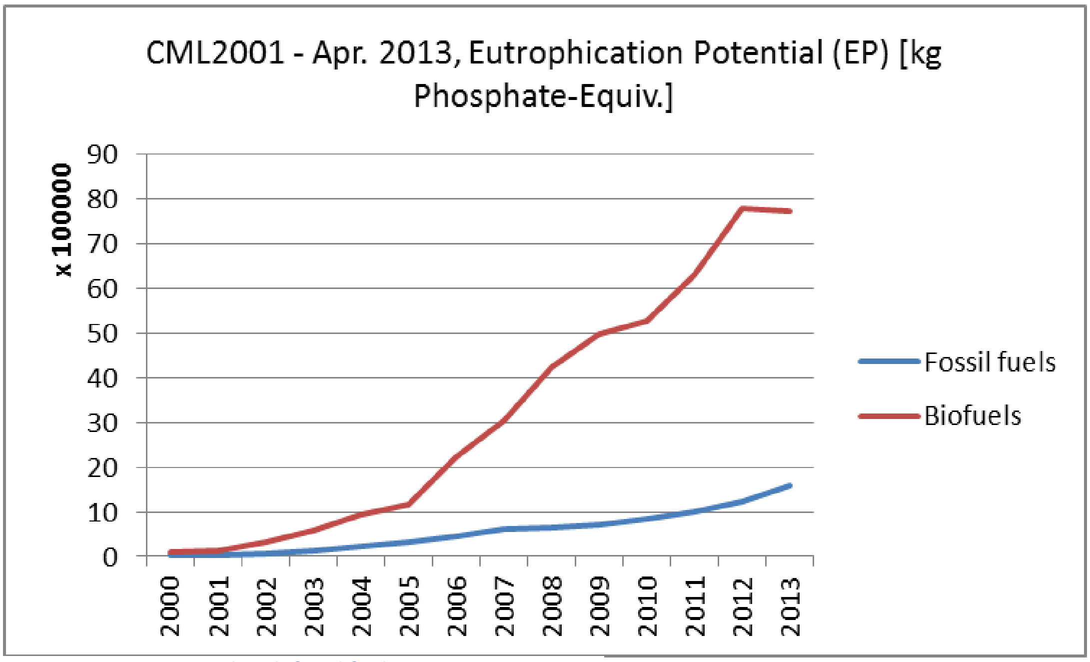


Figure 27: EP compared with fossil fuels

5.4 Freshwater ecotoxicity emissions for the total biofuel consumption

Figure 28 shows the assessed impacts of FAEEP from 2000-2013. Not all datasets indicates impact on freshwater toxicity, there is a lack of complete LCI data. Freshwater toxicity is a local impact. There are two regions that receive almost all the toxicity potential: EU and the other countries. This figure should be used with care: there are only some of the datasets that indicates impacts on toxicity (see Section 5.6.7). In other countries it’s corn for ethanol and palm oil that are influencing the most. In the EU it’s the rapeseed agriculture and ethanol grain agriculture. A comparison with toxicity compared to the fossil fuels would be interesting, but no studies are reporting this impact category.

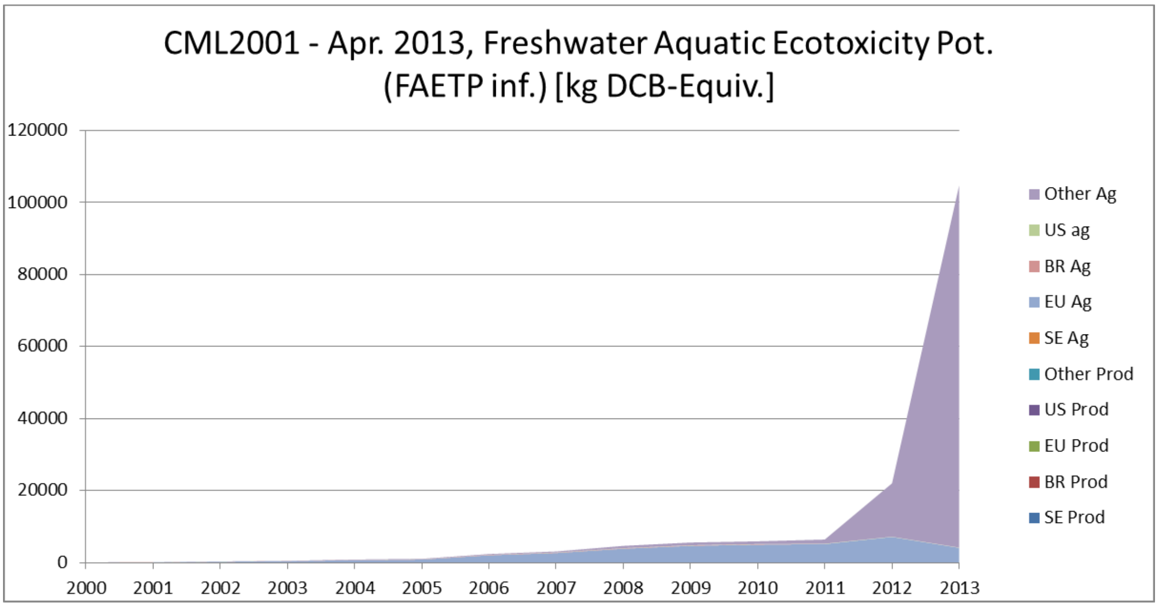


Figure 28: Freshwater ecotoxicity potential for biofuels.

5.5 Photochemical ozone creation emissions for the total biofuel consumption

The potential smog creation potential of kg ethene eqvivalents was plotted against time for feedstock and fuel production of the fuels in each region (see Figure 29). The POCP impact is mostly affected by the amount sugarcane from ethanol consumed. The sugarcane ethanol has large impacts of POCP compared to the other fuels (see Section 5.6.8). The EU agriculture impacts from EU wheat and rapeseed. In the Swedish agriculture it’s only the rapeseed that causes POCP impact. The increase in the year from 2012 to 2013 is the result of an increase of sugarcane as feedstock, the sugarcane ethanol are not imported from Brazil but from other South American countries.

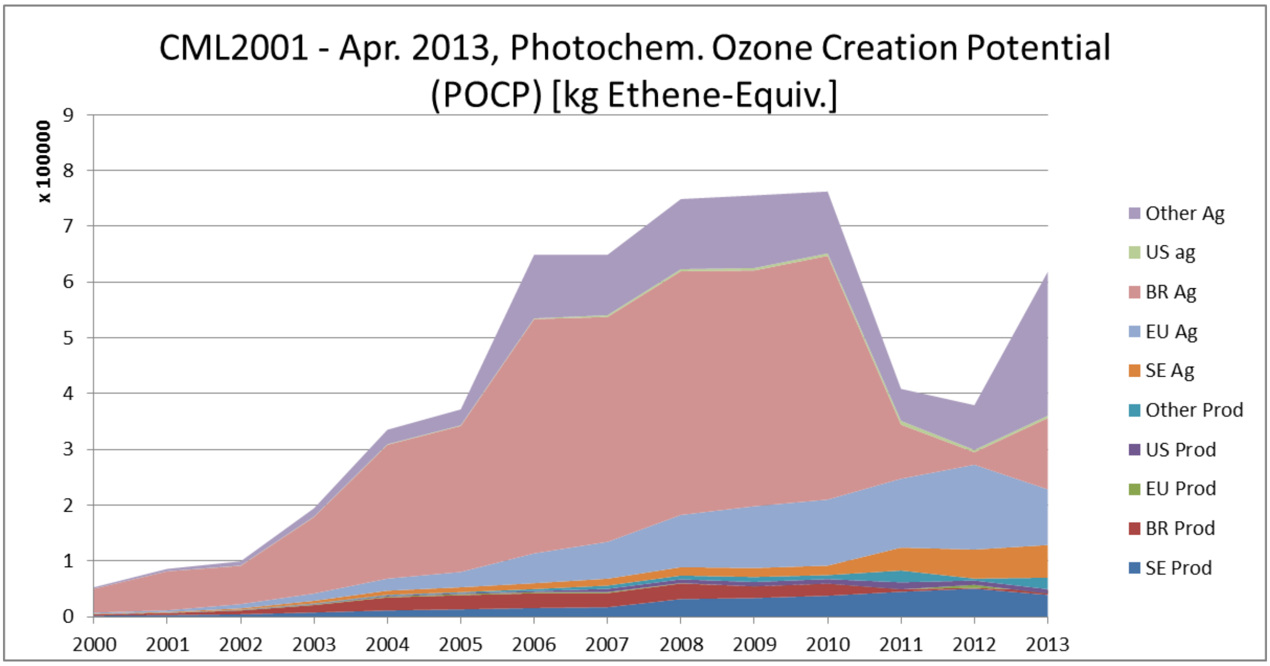


Figure 29: POCP for biofuels.

The POCP emission is, as described in the previous section, in close correlation to the utilization of sugarcane ethanol. From 2011 the assessed impact is almost the same as for fossil fuels (see Figure 29).

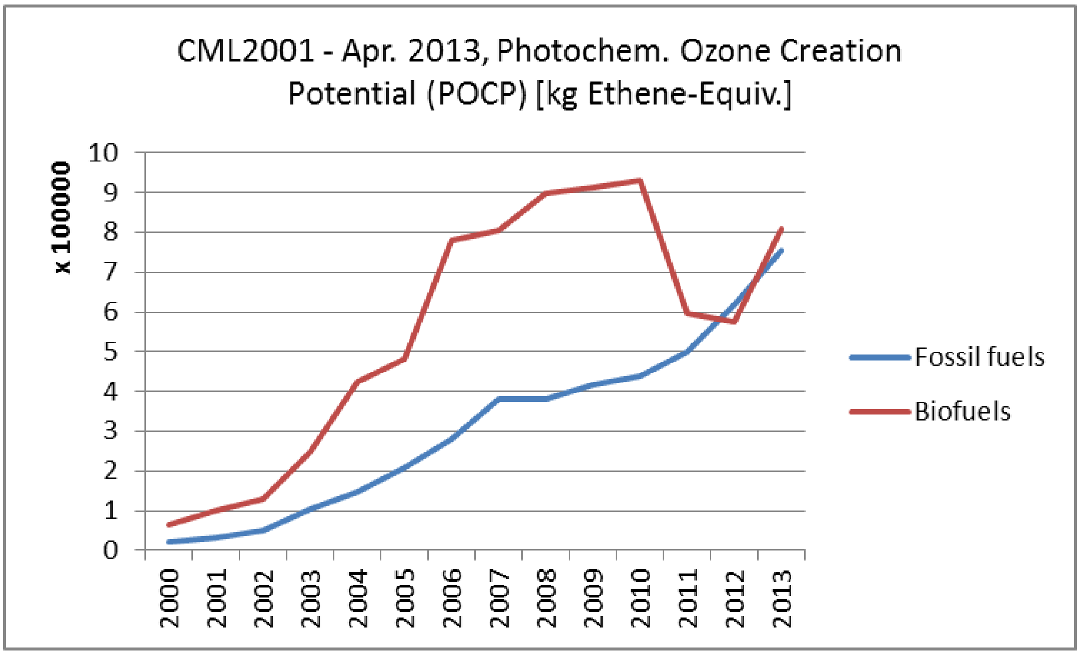


Figure 29: POCP compared with fossil fuels.

5.6 Analysis of the accumulated impact life cycle assessment

The results of the accumulated impacts are interpreted in the following sections. At first, the fuels are analysed on their impact contribution per MJ of fuel and per Kg of feedstock. The analysis further consists of a comparison with the results of other studies.

5.6.1 Global warming potential per fuel and feedstock

The GWP range from 3-100g CO2 eqv. per MJ fuels (see Figure 30). The lowest value, 3g CO_2 eqv. is for HVO produced from animal waste. This dataset has no input of feedstock as animal waste is assumed to be a rest product. EU ethanol from grains (wheat, barley, rye and triticale), HVO from rapeseed, RME and ethanol from corn are by these datasets relatively high emitters. As can be seen in Figure 30 HVO from waste, Brazilian ethanol and Swedish wheat ethanol gives low contribution to GWP-eqv. per MJ fuel. Hot spots for the feedstock datasets are rape seed for HVO (named rape seed 2 in the Figure 31) and RME, ethanol grains, animal fat and corn. Rapeseed, corn and ethanol grains gives high values on GWP since the agriculture step releases NO_2 that corresponds to around 60% of the GWP emissions and CO_2 40%. Sugarcane, palm oil and sugar beets release a small amount of GWP. The values per datasets are greatly influencing the overall GWP impact.

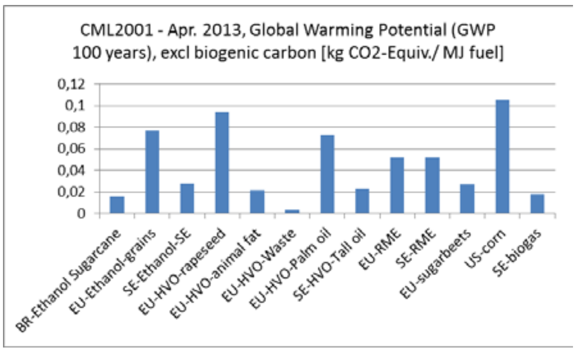


Figure 30 GWP/MJ fuel

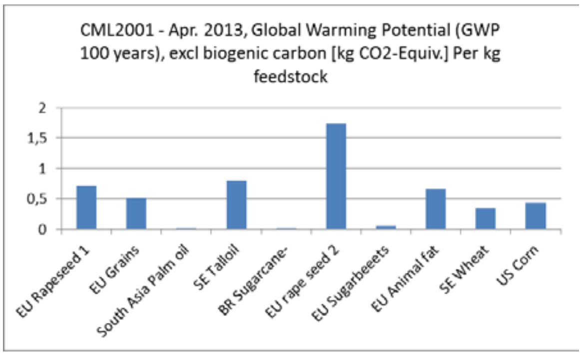


Figure 31 GWP/Kg feedstock

5.6.2 Comparison of global warming potential with other studies

Figure 32 gives a comparison between the used GWP values compared to other studies. The RME agriculture and production is a large contributor for the accumulated impact.

A comparison is obtained with data from Börjesson et. al. (2010) and Bernesson et al. (2006) (see Figure 32). Both of the reference values are based on energy allocations. Börjesson et al. (2010) has grain agriculture as a reference scenario and from Bernesson et al. (2006) values for large scale RME production are considered. The used value for FAME production is not allocated, therefore the RME indicates on higher values in all those impact categories.

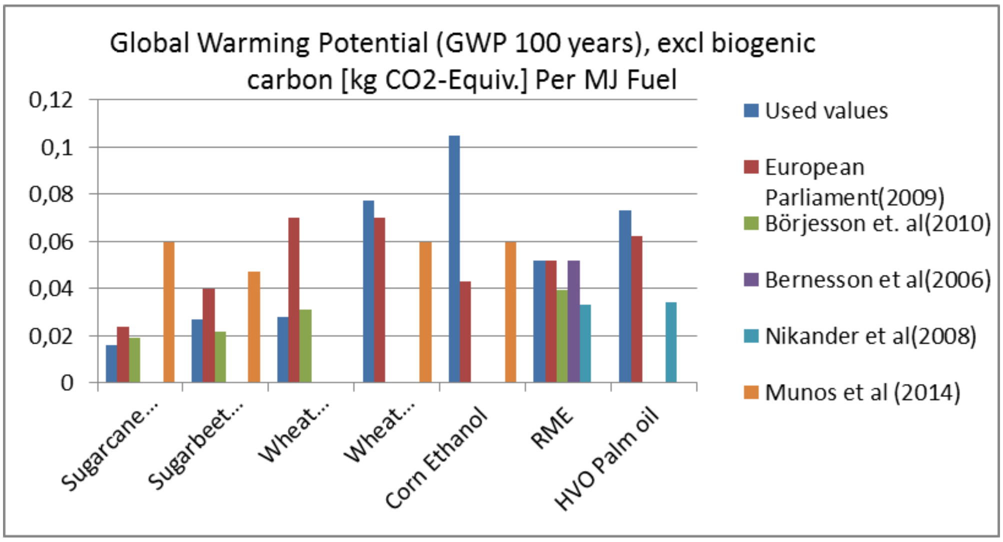
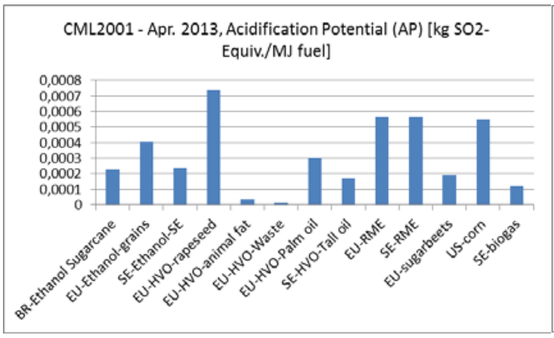


Figure 32 Global warming potential compared to other studies.

5.6.3 Acidification potential per MJ fuel and per kg feedstock

The AP of the different biofuels shows large differences (see Figure 33). The range is from a very low level per MJ of fuel, 0.0012g SO_2 eqv. for EU HVO produced from animal waste to 0.74 g SO_2 eqv. for HVO produced from rapeseed feedstock. The highest emitter of acidification causing emissions is HVO produced from rapeseed, RME and corn produced ethanol. The HVO from rapeseed gives emissions from different combustion steps by the release of SO_2. The RME production also implies combustion of fossil fuels, utilizes phosphoric acid and methanol which release SO_2 emissions. Corn production uses significant amounts of energy, and US electricity mix and transportation is responsible for a large part of the SO_2 eqv. The production process of HVO from tall oil and palm oil is mainly emitting SO_2 eqv. in the processing of the fuels where transportation and natural gas utilization are hotspots.

The AP is high for production of rapeseed, grains for ethanol and corn (see Figure 34). All of those seeds contributes to high impact because of their release of ammonia, nitrogen oxides, nitrogen monoxide and sulphur dioxide.



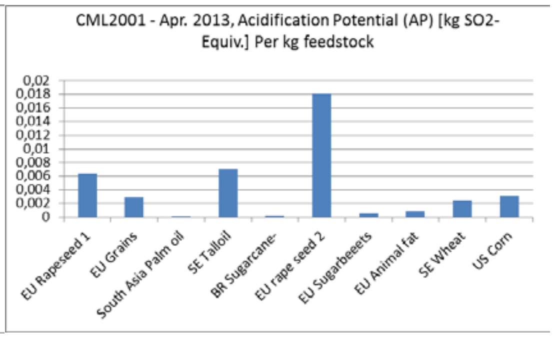


Figure 33: AP per MJ fuel

Figure 34: AP per kg feedstock

5.6.4 Comparison of acidification potential with other studies

The comparison of AP is done with values from Börjesson et al. (2010) calculated on energy allocation, Bernesson et al. (2006) for RME, energy allocation and Munoz et al. (2014) that used economic allocation (see Figure 35). This comparison also shows a large deviation in the results of LCAs (see Figure 24). The largest deviation is the AP for the used value of rapeseed compared with Börjesson et al. (2010) and Bernesson et al. (2006), the used value is not allocated and the modelling is for European conditions. Another interesting outcome from the comparison is the variation between the used LCI data from (Martin et al., 2014) compared to the LCIA result from Börjesson et al. (2010) both for ethanol from Swedish wheat.

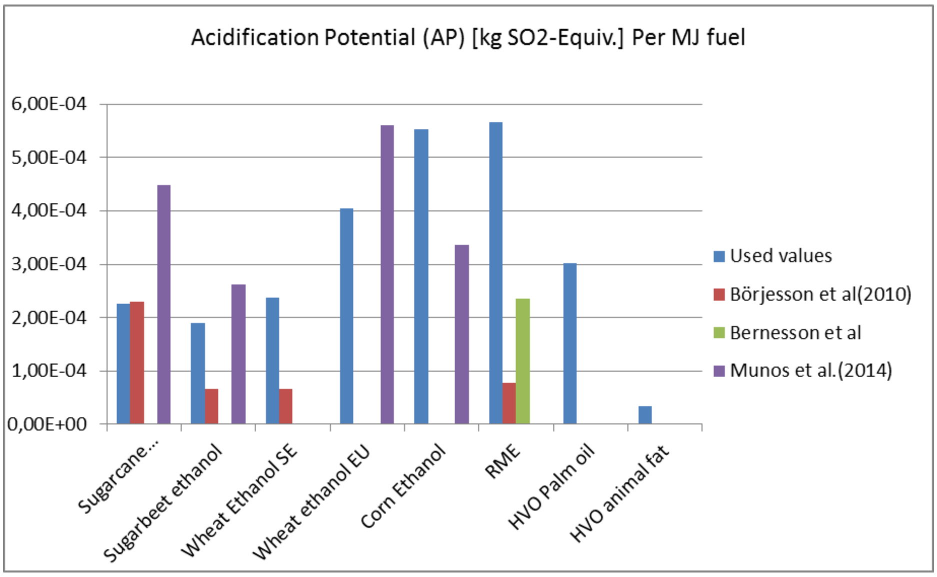
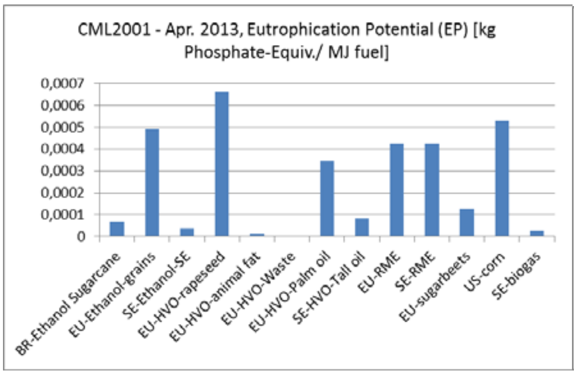


Figure 35: AP comparison with other studies.

5.6.5 Eutrophication potential per fuel and feedstock

he eutrophication potential is high for EU ethanol, EU HVO rapeseed, RME and corn ethanol. For all of those fuels it's the agriculture that is mainly causing eutrophication emissions (see Figure 36 and 37)



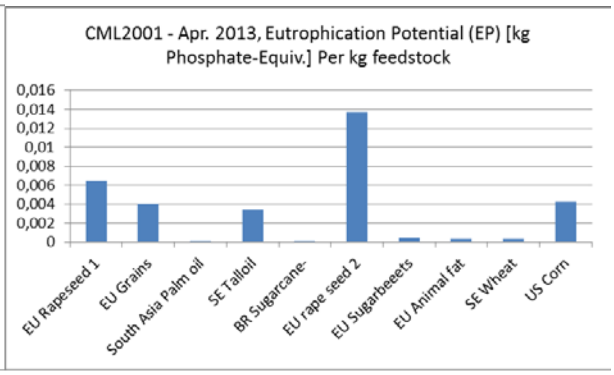


Figure 36: EP per MJ fuel

Figure 37: EP per kg feedstock

5.6.6 Comparison of eutrophication potential with other studies

In contrast with studies from Börjesson et al. (2010) and Bernesson et al. (2006), sugarcane EP is similar whereas wheat ethanol is lower and RME higher for the used datasets (see Figure 38).

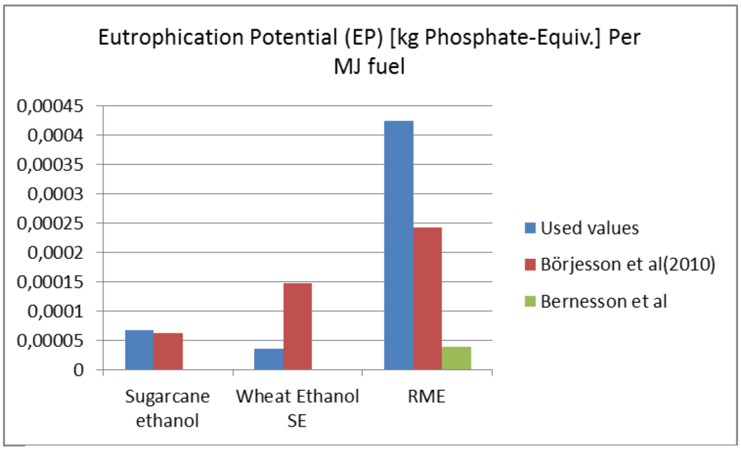
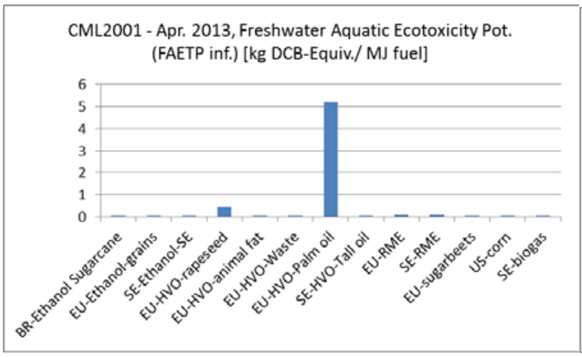


Figure 38: EP compared with other studies.

5.6.7 Freshwater ecotoxicity per fuel and feedstock

Pesticides in Swedish agriculture is not included in the Swedish ethanol dataset, compared with the EU ethanol, the relative pesticide use is low (see Figure 39). The datasets of palm oil and rape seed is by far the most contributing feedstock for freshwater aquatic Eco toxicity (see Figure 40). Rape seed 1 is used for RME and rapeseed 2 for HVO production. Various pesticides are used in the irrigation phase of rape seed and palm oil. No other LCA study indicates the impact of toxicity potential therefore, no comparison is made.



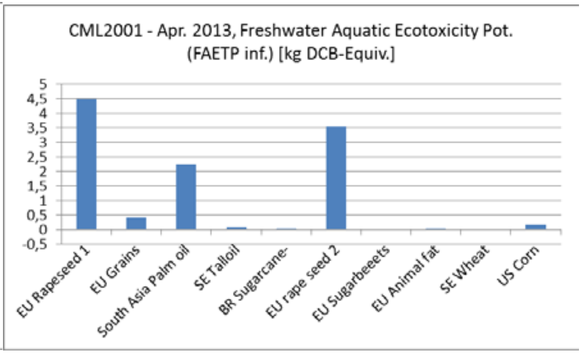
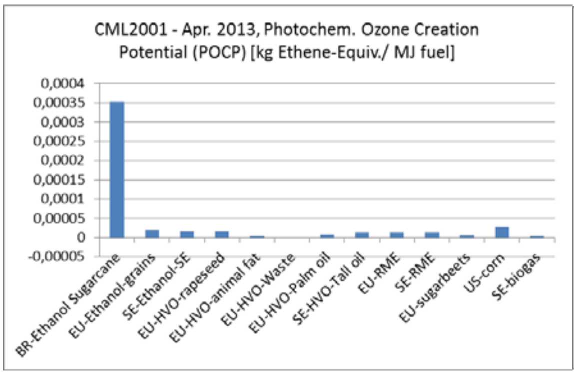


Figure 39 FAEP per MJ fuel

Figure 40 FAEP per kg feedstock.

5.6.8 Photochemical ozone creation potential per fuel and feedstock

The contribution of POCP is shown in Figure 41 for well to gate per MJ fuel and per kg feedstock in figure 42. The POCP-equivalents shows on large differences between the feedstock. A significant change of outcome is seen in the rapeseed datasets, the rapeseed 2 for HVO production and rapeseed 1 for RME production are completely different. At the same time the agriculture would be slightly similar. The negative value of the rapeseed for RME is possible as the dataset is modelled with uptake of nitrogen monoxide. The rape seed data set 2 dataset used for HVO does not have any uptake of nitrogen monoxide and is releasing non-methane volatile organic compounds and nitrogen oxides. For sugarcane, 98% of the POCP is caused by carbon monoxide emissions.



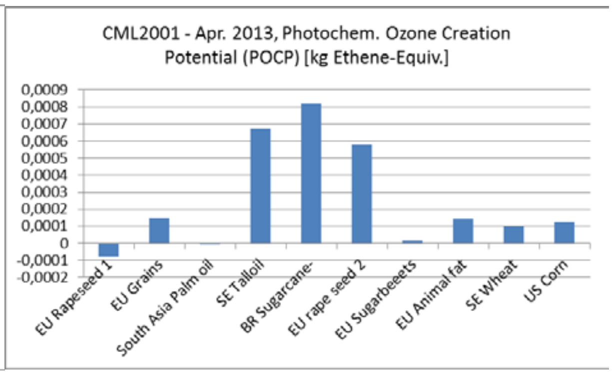


Figure: 41 POCP per MJ fuel.

Figure 42: POCP per kg feedstock.

5.6.8 Comparison of photo-chemical ozone creation potential with other studies

When the assessed values of POCP are compared with others' results the variance is large (see Figure 43). In contrast to Börjesson et al. (2010) the used values are high, but not in relation to Munoz et al. (2010).

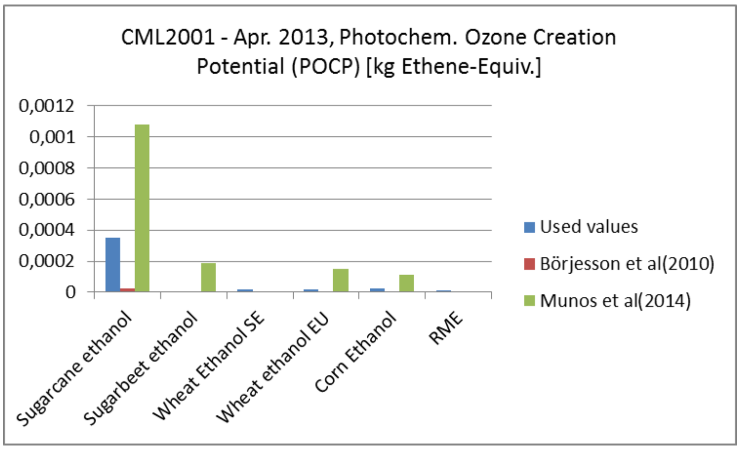


Figure 43: POCP compared with other studies.

5.6.9 Uncertainties in the results

Often a sensitivity analysis is fundamental for the LCA. In this case, all values of the datasets presented and corresponding accumulated impact can only be seen as approximations. A successful sensitivity analysis could be made if the datasets were modelled with ability to change the allocation factors. For the datasets used, the allocation procedures are different. Many of the datasets can't be edited since they are an aggregated process, which implies that all elementary flows are merged together and therefore the processes can't be modelled with different allocations. The datasets in this report is only approximations: for most of the datasets there are large uncertainties in the LCIs. Another source of uncertainty is that the aggregated impacts are based upon assumptions. Only the years from 2011-2013 are cross checked with amount of imported goods per each region. It is often hard to get a comprehensive data collection of toxicity impacts, this has been experienced as almost none of the used datasets gave influence on aquatic ecotoxicity. The dilemma of large LCI gaps in toxicity substances have been highlighted by e.g. Finnveden (2000).

The fossil fuels datasets were used as approximations. The fossil fuels can be modelled by their crude oil origin according to Figure 15. If done in the more precise manner, the regional extraction and production emissions could be studies.

6 Discussion

The discussion is divided into a discussion on the systematic literature review of the Swedish ESA environmental categories and a discussion for the accumulated LCA for the biofuel consumption. There will also be a part about the implications of a limited scope.

6.1 Discussion of meta-study

From the meta-study the impact assessed in Swedish biofuel research was outlined. There is a clear dominance of GWP and energy assessments in the research (see figure 12). Only 11 of the interpreted studies motivate their environmental scope. Of those, 4 states that GWP or CO_2 is the most significant impact from biofuels. The number of studies including more than three environmental categories of environmental importance was only 22%. The chosen impacts should always be clearly stated in the goal and scope definition with motivation, something that surprisingly not was the normal occasion. Even if the assessments were not regulated by the ISO standard, proxy should always be to motivate the selection of impact categories. Although global warming is an effect of large importance in the transport sector, other impacts can't be foreseen. If the environmental performance of a biofuel is investigated, there are many impacts that are relevant more than global warming. In LCA community there is an acceptance that areas of protection are human health, natural environment, natural resources and parts of the manmade environment, impacts on those field is the ground for the assessments (Haes et al., 1999). The ISO standard for LCAs states that "Selection of impact categories and classification involves identification of the categories of environmental impacts which is of relevance to the study" (Klöpffer and Grahl, 2014). The ESAs have no mandatory framework but should motivate theirs' procedure for selection.

6.1.1Why are the assessed impacts limited?

For a majority of the analyses GWP is the core of the environmental impacts in aim, scope, discussion and conclusion. The reason for the GWP focus might be several: policy pressures from the European Union giving requirements on only GWP savings compared to fossil fuels with increased demands, the Swedish vision of fossil free vehicle fleet in 2050, adapted by the Swedish parliament (SOU, 2013). Sustainability criteria are given by the EU and implemented in Swedish law with high demands on carbon performance and little about other consequences. There is also an ongoing awareness of carbon emissions from organizations like IPCC and IEA and the world's leaders struggle to get a new more comprehensive climate agreement of the Kyoto protocol. Carbon emissions are important but the other impacts can´t be ignored and could be better included in life cycle assessments. Another hypothesis is the undeveloped methods and the hardship to find LCI data, especially soil characteristics and biodiversity are hard to assess, the impacts are highly dependent on local conditions and there are no accepted characterization methods which can be one other possible reason for exclusion.

6.1.2 Other important aspects

Not all types of impacts are well covered in a LCA, the methods of land use including biodiversity and resource aspects, including fresh water are problematic and needs further improvements (Finnveden et al., 2009). There is often a conflict between in depth information and applicability (Ny, Macdonald, Yamamoto and Rob, 2006). LCA has been criticized for its low consideration with local characteristics. LCA is insufficient for measuring changes in biodiversity, only few indicators cover impacts on biodiversity such as acidification, eutrophication and toxicity. Indirect land use changes are either not included in the analyses (Bicalho, Jaques and Cecile, 2013). Rebound effects, ILUC and market mechanisms can be included in consequential LCAs but they require scenario modelling (Ibid.). There are new developed methods to estimate biodiversity loss, yet those assessments are not complete and further research is needed. They will however not be accurate since there are large variances geographically in ecosystems (Ibid.). Further problems occur in the agriculture practice; due to insufficient and uncertain data on acidification and toxicity (Finnveden, 2000). To get more precise estimations of the consequences primary data for the area can significantly increase the accuracy, on the other hand, there is an optimum between the effort to collect primary data and to use the existing secondary data, it is also problematic that most of the databases only gives European data (Bicalho, Jaques and Cecile, 2013).

6.2 Implications with to narrow scope

Since many studies analyzed in the META study have a narrow scope on the assessed impacts this may contradict the core principle of LCA methodology: to avoid problem shifting from different life cycles, regions and between environmental problems. Finnveden et al. (2009) recommends that many indicators shall be used and compared against each other to avoid bias and give recommendation based on the best choice from various impacts. Yang et al. (2012) notes the importance to include agriculture related impacts e.g. fertilizers and pesticide use. Wiloso et al. (2012 and Hansen et al. (2013) have also mentioned problems for biofuel LCAs to include eutrophication, acidification, toxicity and land use, water use and biodiversity in the assessments. Bai, Luo and Van Der Voet (2010) draws similar conclusions as Yang et al. (2012) from a study of ethanol produced from switch grass, showing that when GHG emissions decreased compared to fossil fuels the environmental pressure has moved to other impact categories, especially human toxicity, eutrophication and photochemical oxidation potential. Hansen et al. (2013) addresses the major challenges of including deforestation with soil erosion, changed carbon stock and albedo effects, the degradation of biodiversity, nutrient removal and leakage, indirect land use changes and social and price mechanisms drawbacks. Sokka (2011) explains that GHG is not the only emission to assess for a full picture for environmental impacts of a LCA. Laurent et al. (2012) warns that simplifications in impact assessment with narrow indicators can mislead policy decisions and cause environmental problem shifting. Norborg et al. (2014) states that the net gain of GHG emission gives environmental burden shifting to the expense of other impacts. Pennington et al. (2004) also warns that inconsistencies in the comparison and lack of appropriate LCI data can contribute to unintended bias.

Even though there is a lack of complete LCAs and environmental understanding among fuels in the Swedish research, there have long been policies and subsidies to the fuels. Can still policymakers give strong subsidies to promote the biofuels? LCA is only an indicating ESA tool and are never assessing the actual emissions, but that there is a potential linkage between the product's, process life cycle or life cycle step and its impact (Heijungs, 2014). Therefore, the LCA can lead to uninformed decisions when the impact categories are limited (Laurent, Olsen and Hauschild, 2012). Not many of the studies are explaining their uncertainties in the assessments, this should carefully be explained for increased transparency of the environmental risks (Pennington et al., 2004). Global warming is crucial to consider for better environmental performance but there are other areas of protection and the decisions should be based on a best available assessment overall.

6.3 Discussion on Sweden's biofuel consumption

Impacts occurring due to Swedish biofuel consumption have been substantial. This can be seen in the comparison with fossil fuels (see figure 28,33,41), even though the datasets for each biofuel can be criticized and eventually replaced by a more developed, and with region specific dataset, the impacts besides GWP are of large scale. The eutrophication potential from biofuels compared with fossil fuel system is 5.9 times larger. For AP and POCP the levels are 1.8 times higher for biofuels. With this comparison in mind it, the potential levels of AP, EP and POCP should always be included in LCAs for biofuels to avoid problem shifting. Also freshwater demand for the biofuels should be investigated.

6.3.1 Origin of the impacts

The regional impact are relatively small for Sweden, the most of the emissions occur in the rest of Europe, Brazil and in other countries. Sweden has exported a lot of emissions to the other countries. Overall the feedstock from Sweden has been around 20%, a bit lower the years from 2007-2010 and 25% in 2012 and then decreased again. Brazil has been dominating the imports of feedstock to 2007 and then EU and other countries have been taken the largest parts. Since 2012 new importation countries have been increasing their import parts (see figure 10).

Overall the emissions are decreasing due to the increased amount of HVO. HVO from tall oil and animal fat and waste, those fuels are giving better performance in EP and GWP than all other fuels. Notice that HVO from rapeseed is not preferable (figure 38). Still, tall oil are emitting relatively large amount of POCP and AP. Nevertheless, the datasets for HVO tall oil, animal fat and animal waste are modelled upon energy balances: there is a possibilyt that the simplification might influence the LCIA results. At the same time, no studies for comparing of HVO from tall oil and animal waste were found, not for production neither for combustion. HVO from palm oil has been consumed in larger amount the years 2012-2013, the toxicity potential are drastically higher than for the other fuels, this could be the actual case or because of data lack for pesticide use among the other fuels, as discussed earlier.

The accumulated LCA was at first supposed to include more impact categories as recommended by Finnveden et al., (2009) and to further assess the impacts from agriculture, highlighted by Yang et al. (2012) and Dalgaard (2008). This was not fully achieved since the datasets shows abnormal and misleading values for toxicity (see Figure 38, 39). The land use results are not presented in the results, the reason is the lack of LCI data for land use and land use change.

6.4 Better practice

This section discusses potential developments for environmental assessments on biofuels.

6.4.1 Robustness and better framework

To compare the biofuels fairly and base the policies on a more complete assessment, two issues need to be considered; lack of data and lack of standards. Those two issues have been identified by Bicalho et al. (2013) as limitations in the EU renewable energy directive on biofuels. The dilemma with data gaps has been a source of error the in the performed LCA. The data gaps are influencing in abundance of datasets for region specific agriculture and production, this issue is also a general limitation for LCAs (Finnveden, 2000). There is a low transparency in the published LCAs LCI step, making it hard to know the exact system boundaries and input output data. The transparency and possibility to modify is also a problem in some of the databases aggregated datasets. The LCA methods for biofuel assessments are not homogenous. For a fair comparison, same functional unit, spatial dimension, time dimension, allocation procedure and environmental impacts should be used (Finnveden, 2000). One potential solution for those issues is to establish a standard for biofuel LCA. It could be done by EPD-method (Environmental product declaration). The EPD has a regulatory framework for different products called PCR (product category rules), the PCR achieves comparable results of the LCAs (Del Borghi, 2013).

6.4.2 Inclusion of indirect land use change consequences

The land use is also important to consider since the agriculture requires large amount of arable lands. Two types of land use interventions are usually considered, land transformation and land occupation. Land transformation occurs when an area are transformed to agriculture land for feedstock agriculture, land occupations is the area of land occupied (Koellner and Geyer, 2013). With a growing demand for biofuels; the pressure on arable land will increase, causing indirect rebound effects of land transformation. Studies have proved that GWP emissions occurring from ILUC can be of highly important range, but the uncertainties and geographical differences are large (Ferreira Filho and Horridge, 2014; Plevin et al., 2010) . Especially the increase of palm oil from Malaysia and Indonesia that accounts for 19% of HVO has been criticized for dramatically negative consequences of ILUC causing deforestation (Gilbert, 2012). Another important aspect is the biodiversity loss from LUC and ILUC. Land use change is a key driver for terrestrial biodiversity loss (Ibid.). There are pioneers working on quantifying those losses, but the uncertainties are large (Souza, Teixeira and Ostermann, 2014).

Another interesting development is the recent research on local conditions and carbon soil stocks in different regions for different feedstock (Staff writers, 2015). So maybe in some years the site specific data methods are available for better assessments.

6.4.3 Avoiding sub-optimized policies

The aim for many policies of biofuels is to increase the consumption of biofuels, any biofuel that reach the GWP target: reduction of at least 30% of GWP compared to fossil fuels and that no highly biodiverse and soil carbon rich area are directly affected. Higher sustainability claims should be introduced in Swedish policy context, both for decreased local environmental harm and to promote better environmental performing fuels.

6.4.4 Biofuels in larger context

Overall, EU's consumption is exceeding its so called environmental cropland footprint. The EU demand rises steady and increases the pressure on the global croplands. EU need to use less land per capita for safe operating space towards the planetary boundaries (O'Brien, Schütz and Bringezu, 2015). Giljum et al. (2008) states that EU's policies lead to less extraction and production of resources domestic. The environmental pressure of EU grows at the same time, but its effects are caused in more resource rich areas (see Figure 20). The safest approach to solve this environmental problem shifts is to apply long term polices with less resource demand. Giljum et al. (2008) argues for a global allocation structure implemented in export and import chains to allocate the environmental impacts correctly according to countries import.

Biofuels can be a dilemma, causing high local emissions, land use and indirect land use change. If the Swedish EPA's goal of solving environmental problems shall be solved without exporting them to other regions, Sweden has to change its biofuel policies. There is also a significant question one can ask, what are the areas of protection? If global warming is the only area, Swedish biofuel strategy has been successful.

The planetary boundaries can act as a framework of highly important environmental areas of protection. Many of the impact categories or environmental areas are connected, but the nitrogen cycle and biodiversity loss have exceeded the boundaries even more than global warming. Land use change is also at a critical levels (Steffen et al., 2015).

7. Conclusions

The objectives of this master thesis were to analyze the environmental impacts assessed in the Swedish biofuel research, to account for the impacts caused from biofuel consumption and to outline the potential implications of limiting impact categories.

One of the findings is that the impact categories in Swedish biofuel ESA research 2000-2015 have been paying much attention to GWP and energy analyses. This can be seen as the scope in the analyses has often been too narrow without explanation. The LCAs gives a bit broader scope of impact categories compared to other ESAs whereas eutrophication, acidification and land use are the most commonly assessed impacts besides of global GWP. Few of the ESAs are basing their conclusions on more impact categories than GWP. Also, the studies considering land use tend to assess GWP emissions from soil or soil organic carbon.

An accumulated well to gate LCA was performed for the years 2000-2013 for Sweden's total biofuel consumption. Included environmental impacts were GWP, AP, EP, POCP and freshwater toxicity. From this LCA emissions could be traced from feedstock and biofuel production processes in different regions. The regions were divided in Sweden, rest of EU, Brazil, USA and other countries. A comparison with full life cycle perspective was made between Sweden's consumption of biofuels compared to a reference scenario based on fossil fuels. This comparison showed that while GWP has been lower, the other assessed impacts were dramatically higher than for fossil fuels. Many of the emissions from the biofuel consumption are local. However, most of those local impacts are emitted outside Sweden, contradicting the environmental agency's goal of solving problems and not export them to other countries.

Problems occurring from the narrow scope in policies and ESA and the increase of emissions in other regions have been discussed. The largest problem for biofuel assessments is lack of site specific data and lack of standards. A more holistic policy is needed with wider scope to reach overall environmental benefits from biofuels. The proposed standard needs to account for ILUC emissions, since the change can be of significant order. Renewable fuels do not imply sustainable fuels: the biofuels need to be assessed on performance in many environmental categories.

Future research

To get higher reliability of those results, more site specific data should be used. A better model, with more updated LCIs, especially for tall oil and animal waste HVO. The assumptions in the flow analysis should be more precise, this is somehow difficult to obtain since the information of biofuels import, export and production is under secrecy. Both direct and indirect land use change are highly important issues and should be assessed further. It is also significant to develop a method about how those issues should be implemented in policy for other impact categories than GWP.

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Appendix 1

Reviewed articles :

			Environmental impact indicators													
Author	**year**	**Type**	**GW**	**En**	**AD**	**AP**	**EP**	**OD**	**POC**	**PS**	**Particl**		**Huma**	**LU/LUC**	**TP**	**WD**
Ahlgren & Lucia	2014	Review Land us	X											LUC-GHG		
Ahlgren et al.	2008	LCA	X	X		X	X							LU		
Ahlgren et al.	2009	LCA	X	X		X	X							LU		
Ahlgren et al.	2013	Critical method	X	X										LUC-gwp,soc,biogenic		
Alfors et al.	2010	Review of proc	X	X										LU+LUC-gwp,		
Andersson & Harvey	2006	System Analysi	x	x												
Arvidsson et al.	2010	LCA	X	X		X	X									
Bauer & Hultenberg	2013	System analysi	X	X												
Bengtsson et al.	2012	LCA	X		X	X	X				X		X			
Bernesson et al.	2006	LCA	X	X		X	X		X							
Bernesson et al.	2004	LCA	X			X	X		X							
Bernstad et Al.	2011	LCA	X	X		X	X	X	X							
Bernstad et Al.	2012	LCA	X	X		X	X									
Bezergianni et al.	2014	LCA,sustainabi	X	X												
Björklund et al.	2001	Energy and env	X	X												
Brau et al.	2013	System Analysi	X	X												
Brynolf et al.	2014	LCA	X	X		X	X			X	X					
Börjesson and Mattiassoon	2008	Review of fuel	X	X										LU, GHG		
Börjesson et al.	2013	System analysi	X	X												
Börjesson et al.	2013	Review	X	X										LUC-GHG		
Börjesson et al.	2013	Scenario study	X	X												
Börjesson et al.	2011	LCA,LUC	X	X			X							LU GHG		
Börjesson et al.	2009	Critical assessn	X	X										LUC GHG		
Caspeta et al.	2013	Energy,econom	X	X												
Ekman et al.	2011	LCA	X	X			X							LUC,GHG		
Ekman et al.	2013	LCA	X	X												
Englund et al.	2012	EIA Recomenda	X											LUC		
Garraín et al.	2014	LCA	X	X												
Gonzales Garcia et al.	2012	LCA	X	X	X	X	X	X	X				X		X	
Gonzalez Garzia et al.	2011	LCA	X	X	X	X	X	X		X			X		X	
Gustavsson & Karlsson	2006	co2- mitigation	X													

			Environmental impact indicators													
Author	**year**	**Type**	**GW**	**En**	**AD**	**AP**	**EP**	**OD**	**POC**	**PS**	**Particl**		**Huma**	**LU/LUC**	**TP**	**WD**
Hagman et al.	2013	LCA;water	X													X
Hansen et al.	2013	Land-use	X											LUC		
Janssen et al.	2014	LCA	X	X		X	X		X							
Jensen et al.	2012	Review c02	X											LUC-GHG		
Joelsson and Gustavsson	2012	LCA	X	X												
Johansson et al.	2014	Comparison ca	X	X												
Karlsson & Börjesson	2013	LCA;summary r	X	X												
Karlsson et al.	2014	LCA	X	X												
Khatiwada et al.	2012	Critical assessm	X											LUC-ghg		
Khatiwada et al.	2011	LCA	X	X												
Kimming et al.	2011	LCA	X	X	X											
Lageveld et al.	2012	Environmental assessment									a			LU		
Lantz et al.	2013	LCA	X											LUC		
Larsolle & Andersson	2013	LCA	X	X												
Lubbe et al.	2012	Scenario analy	X													
Luterbacher et al.	2009	LCA	X	X												
Lutherbacher et al.	2009	LCA	X	X												
Margeot et al.	2009	Review	X	X												
Martin et al.	2014	LCA	X	X		X	X									
Moghaddam et al.	2013	System analysi	X	X												
Naqvi et al.	2013	System Analysi	X	X												
Nordborg et al.	2014	Ecotoxicity assessment													X	
Pettersson and Harvey	2012	System Analysi	X	X												
Risen et al.	2013	System Analysi	X	X												
Samiei and Fröling	2014	System analysi	X		X	X			X							
Sparovek et al.	2008	Critical assessment												LUC-area		
Suer et al.	2011	LCA	X	X		X	X		X				X	LU	X	
Tidåker et al.	2011	LCA	X	X		X	X							LUC- ghg, nutrients		
Tufvesson et al.	2013	LCA	X	X		X	X									
Tufvesson et al.	2013	LCA	X	X												
Wang et al.	2013	LCA	X	X										LU		
Wang et al.	2012	Environmental	X	X			X									
Wetterlund et al.	2012	System Analysi	X	X												

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Appendix 2

Ethanol feedstock							
	2000	2001	2002	2003	2004	2005	2006
Wheat	30%	30%	30%	30%	30%	30%	30%
Corn	5%	5%	5%	5%	5%	5%	5%
Sugarcane	48%	48%	48%	48%	48%	48%	48%
Triticale	4%	4%	4%	4%	4%	4%	4%
Barley	5%	5%	5%	5%	5%	5%	5%
Sugarbeet	4%	4%	4%	4%	4%	4%	4%
Rye	3%	3%	3%	3%	3%	3%	3%
Molasses	1%	1%	1%	1%	1%	1%	1%
	2007	2008	2009	2010	2011	2012	2013
Wheat	32%	35%	35%	40%	45%	53%	35%
Corn	12%	12%	15%	15%	31%	30%	24%
Sugarcane	43%	42%	39%	33%	7%	3%	17%
Triticale	3%	3%	3%	3%	2%	5%	15%
Barley	3%	3%	3%	3%	7%	5%	4%
Sugarbeet	3%	2%	2%	2%	3%	2%	4%
Rye	3%	2%	2%	3%	5%	2%	1%
Molasses	1%	1%	1%	1%	0%	0%	0%

Appendix 3

	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
Wheat SE	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	52%	45%
Wheat EU	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	48%	55%
Corn US	40%	40%	40%	40%	40%	40%	40%	40%	40%	40%	40%	20%	20%	20%
Corn EU	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	10%	70%	10%
Corn Others	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	70%	10%	70%
Sugarcane Br	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	50%	42%
Sugarcane Others	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	50%	58%
Triticale SE	45%	45%	45%	45%	45%	45%	45%	45%	45%	45%	45%	45%	50%	48%
Triticale EU	45%	45%	45%	45%	45%	45%	45%	45%	45%	45%	45%	45%	50%	50%
Triticale Others	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	0%	2%
Barley SE	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	60%	45%
Barley EU	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%
Barley Other	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	10%	25%
Sugarbeets EU	60%	60%	60%	60%	60%	60%	60%	60%	60%	60%	60%	60%	25%	75%
Sugarbeets Others	0%	0%	40%	40%	40%	40%	40%	40%	40%	40%	40%	40%	75%	25%
Rye SE	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	55%	45%
Rye EU	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%	45%	55%
Molasses EU	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%	70%
Molasses Other	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%	30%
Wine Residues EU	80%	80%	80%	80%	80%	80%	80%	80%	80%	80%	80%	80%	80%	80%
Wine Residues Othe	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%	20%

Appendix 4

Ethanol-Produced in Sweden							
SE	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
Wheat-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Triticale-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Barley-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Rye-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
EU							
Wheat-EU	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%
Triticale-EU	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%
Barley-EU	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%
Rye-EU	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%
Other							
Wheat-Other	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%
Triticale-Other	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%
Barley-Other	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%	0,00%
SE	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Wheat-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Triticale-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Barley-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Rye-SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
EU							
Wheat-EU	0,00%	60,00%	60,00%	60,00%	60,00%	60,00%	60,00%
Triticale-EU	0,00%	60,00%	60,00%	60,00%	60,00%	60,00%	60,00%
Barley-EU	0,00%	60,00%	60,00%	60,00%	60,00%	60,00%	60,00%
Rye-EU	0,00%	60,00%	60,00%	60,00%	60,00%	60,00%	60,00%
Other							
Wheat-Other	0,00%	60,00%	60,00%	60,00%	60,00%	60,00%	60,00%
Triticale-Other	0,00%	60,00%	60,00%	60,00%	60,00%	60,00%	60,00%
Barley-Other	0,00%	60,00%	60,00%	60,00%	60,00%	60,00%	60,00%
Produced in Brazil							
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
Sugarcane-Br	100%	100%	100%	100%	100%	100%	100%
Sugarcane-Other	100%	100%	100%	100%	100%	100%	30%
	2007	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Sugarcane-Br	100%	100%	100%	100%	100%	100%	100%
Sugarcane-Other	30%	30%	30%	30%	30%	30%	30%
Ethanol-Produced in US							
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
Corn-US	100%	100%	100%	100%	100%	100%	100%

Corn-Other	20%	20%	20%	20%	20%	20%	20%
	2007	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Corn-US	100%	100%	100%	100%	100%	100%	100%
Corn-Other	20%	20%	20%	20%	20%	20%	20%
Ethanol-Produced in other countries							
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
Sugarcane-Other	70%	70%	70%	70%	70%	70%	70%
Corn-Other	80%	80%	80%	80%	80%	80%	80%
Sugarbeets-other	100%	100%	100%	100%	100%	100%	100%
	2007	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Sugarcane-Other	70%	70%	70%	70%	70%	70%	70%
Corn-Other	80%	80%	80%	80%	80%	80%	80%
Sugarbeets-other	100%	100%	100%	100%	100%	100%	100%

Appendix 5

Origin of FAME feedstock							
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
Australia	0%	0%	0%	0%	0%	0%	0%
EU	85%	85%	85%	85%	85%	85%	85%
SE	2%	2%	2%	2%	2%	2%	2%
Other	13%	13%	13%	13%	13%	13%	13%
	2007	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Australia	0%	0%	0%	0%	0	7%	20%
EU	85%	85%	85%	85%	80%	65%	58%
SE	2%	2%	2%	2%	2%	4%	5%
Other	13%	13%	13%	13%	17%	25%	17%

Production of FAME-Sweden							
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
rape AU	0,00%	0,00%	70,00%	70,00%	70,00%	70,00%	70,00%
Rape EU	0,00%	0,00%	30,00%	30,00%	30,00%	30,00%	30,00%
Rape SE	0,00%	0,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Rape others	0,00%	0,00%	50,00%	50,00%	50,00%	50,00%	50,00%
	2007	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
rape AU	70,00%	70,00%	70,00%	70,00%	70,00%	70,00%	70,00%
Rape EU	30,00%	30,00%	30,00%	30,00%	30,00%	30,00%	30,00%
Rape SE	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%	100,00%
Rape others	50,00%	50,00%	50,00%	50,00%	50,00%	50,00%	50,00%
Production of FAME-EU							
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
rape AU			30,00%	30,00%	30,00%	30,00%	30,00%
Rape EU			70,00%	70,00%	70,00%	70,00%	70,00%
Rape SE			0	0	0	0	0
Rape others			50	50	50	50	50
	2007	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
rape AU	30,00%	30,00%	30,00%	30,00%	30,00%	30,00%	30,00%
Rape EU	70,00%	70,00%	70,00%	70,00%	70,00%	70,00%	70,00%
Rape SE	0	0	0	0	0	0	0
Rape others	50%	50%	50%	50%	50%	50%	50%

Appendix 6

Feedstock for HVO							
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**
Animal waste	**0%**	**0%**	**0%**	**0%**	**0%**	**0%**	**0%**
Talloil	0%	0%	0%	0%	0%	0%	0%
Animal fat	0%	0%	0%	0%	0%	0%	0%
Waste	**0%**	**0%**	**0%**	**0%**	**0%**	**0%**	**0%**
Vegetable fat	**0%**	**0%**	**0%**	**0%**	**0%**	**0%**	**0%**
	2007	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Waste	**0%**	**0%**	**0%**	**0%**	**16%**	**0%**	**51%**
Talloil	0%	0%	0%	0%	84%	**44%**	**26%**
Animal fat	0%	0%	0%	0%	**0%**	**24%**	**19%**
Waste	**0%**	**0%**	**0%**	**0%**	0%	**21%**	**4%**
Vegetable fat	**0%**	**0%**	**0%**	**0%**	0%	**0%**	**0%**
Palm oil	**0%**	**0%**	**0%**	**0%**	**0%**	**10%**	**19%**

Appendix 7

		2000	2001	2002	2003	2004	2005	2006
Diesel	1000m3	2529	2550	2925	3063	3394	3591	3718
Bio blend	1000m3	0	0	4	5	9	9	56
Only diesel	1000m3	2529	2550	2921	3058	3385	3582	3662
Gasoline	1000m3	5335	5381	5463	5494	5439	5414	5311
Ethanol blend	1000m3	0	24	58	125	235	252	248
Only Gasoline	1000m3	5335	5357	5405	5369	5204	5162	5063
%diesel		0,337913	0,338843	0,367829	0,380123	0,411875	0,427625	0,4378
%gasoline		0,662087	0,661157	0,632171	0,619877	0,588125	0,572375	0,5622
		2007	2008	2009	2010	2011	2012	2013
Diesel	1000m3	3902	4020	3956	4330	4486	4454	5125
Bio blend	1000m3	125	160	194	207	269	362	530
Only diesel	1000m3	3777	3860	3762	4123	4217	4092	4595
Gasoline	1000m3	5237	4832	4749	4453	4119	3784	3715
Ethanol blend	1000m3	244	228	229	216	204	191	179
Only Gasoline	1000m3	4993	4604	4520	4237	3915	3593	3536
%diesel		0,448867	0,474422	0,472601	0,511643	0,536974	0,5508	0,583177
%gasoline		0,551133	0,525578	0,527399	0,488357	0,463026	0,4492	0,416823

Sunpine process:

Sunpine energy use		
	E[GJ]	Allocated to raw tall diesel
Methanol	14566	10464,62
electricity	3077	2210,603
Energyuse process	130009	93402,09
Allocated to Beck oil	441100	

The energy use in the sunpine step is from (Barr, 2010). The values are for 100 000 tonnes of hvo. The allocation factor is between raw tall diesel that is produced in the process and beck oil. By dividing the allocated tall oil energy with the total amount the allocation factor for raw tall diesel was given.The allocation factor was 0,72. The energy use in the processes was for burning of tall oil for heat purposes. Thereby the emissions from the Sunpine transesterification need to include burning of tall oil. Numbers for the tall oil emissions was taken from (Gode, Martinsson, Hagberg and Palm, 2011)(table 5). The consumption of 0,72kg tall diesel requires 1kg raw tall oil(Barr, 2010)

Tabel 1 Burning of tall oil (Gode et al., 2011)

Emission	g/MJ fuel
Carbon dioxide	0
Methane	2,00E-03
Dinitrogen oxide	6,00E-04
nitrous oxide	1,60E-02
sulphur dioxide	1,10E-01
particles	4,20E-03

The methanol dataset was taken from PE-international based on atomic weight in the reaction between raw tall oil and methanol to form raw tall diesel.

$$R - COOH + CH_3OH \rightarrow R - COOCH_3 + H_2O$$

SunPine RTD

Fatty acid and resin acids structures

	Mw	Conc.% by weight	Formula	Structure formula
Palmitic	256	5	$C_{16}H_{32}O_2$	$CH_3(CH_2)_{14}COOH$
Stearic	284	5	$C_{18}H_{36}O_2$	$CH_3(CH_2)_{16}COOH$
Oleic	282	25	$C_{18}H_{34}O_2$	$CH_3(CH_2)_7\mathbf{CH{=}CH}(CH_2)_7COOH$
Linoleic	280	40	$C_{18}H_{32}O_2$	$CH_3(CH_2)_4\mathbf{CH{=}CH}CH_2\mathbf{CH{=}CH}(CH_2)_7COOH$
Linolenic	278	5	$C_{18}H_{30}O_2$	$CH_3CH_2\mathbf{CH{=}CH}CH_2\mathbf{CH{=}CH}CH_2\mathbf{CH{=}CH}(CH_2)_7COOH$
Other FA		**Balance**		$C_{12} - C_{24}$ saturated and unsaturated FA
Resin acids	302	**1- 20**	$C_{20}H_{30}O_2$	$(CH_3)_4C_{15}H_{17}COOH$

The R-group is corresponding to the fatty acids presented in figure (x) The mean value of the molecular weight was 280 and for methanol it is 32. Therefore the relationships is 280:32. And for each reaction of fatty acid 0,114285714 parts of methanol are needed.

The modelling of the tall diesel production was recalculated per kg talloil, the input output of the process are shown in table xx.

Inputs			
Electricity [Electric power]	Energy (net calorific value)	0,002277	MJ
Methanol [Organic intermediate products]	Mass	0,003085	kg
raw tall oil [Exjobb]	Mass	1	kg
outputs			
tall oil to refinery [Exjobb]	Mass	0,71	kg
Dust (PM2,5 - PM10) [Particles to air]	Mass	3,92E-07	kg
Methane [Organic emissions to air (group VOC)]	Mass	1,87E-07	kg
Nitrogen oxides [Inorganic emissions to fresh water]	Mass	1,49E-06	kg
Nitrous oxide (laughing gas) [Inorganic emissions to air]	Mass	5,60E-08	kg
Sulphur dioxide [Inorganic emissions to air]	Mass	1,27E-05	kg

Preem hydrofication

The energy for Preem's hydrofication from tall diesel to hvo requires naphta, hydrogen, electricity and heat. Light petroleum gas and burning gas are energy allocated from the HVO with factor 0,89. The energy assessment table are presented in table xy.

Tabell 2 Preem hydrofication aggregated energy

Preem energy use		
	GJ	Allocated to HVO[GJ]
Aggregated Naphta	92655	82781,78
Aggregated hydrogen	17457	15596,8
electricity	381	340,401
energy process use	4429	3957,05
Allocated to LPG	14954	

The energy used in the process are LPG, as for the sunpine process values of emissions was given by "miljöhandboken" (Gode et al., 2011) for light petroleum gas.

Tabell 3 Air emissions for burning of LPG (Gode et al., 2011)

	g/MJ
Carbon dioxide	6,7
Methane	4,61E-02
Lauging gas	3,59E-03
Carbon monoxide	9,14E-03
nitrious ocide	1,56E-02
Suplpur dioxide	2,26E-02
VOC	2,09E-02
Particles	1,28E-04

The inputs of naphta and energy were substituted with Ecoinvents dataset for those processes. Finally, the i/o table of the preem process can be seen in table xx, recalculated per kg of HVO.

Input			
Electricity [Electric power]	Energy (net calorific value)	0,001576	MJ
Naphtha [Organic intermediate products]	Mass	0,001643	kg
tall oil to refinery [Exjobb]	Mass	1	kg
Output			
hvo tall oil [Exjobb]	Mass	0,88	kg
Carbon dioxide [Inorganic emissions to air]	Mass	0,000122	kg
Carbon monoxide [Inorganic emissions to air]	Mass	1,60E-07	kg
Dust (PM2,5 - PM10) [Particles to air]	Mass	2,34E-09	kg
Methane [Organic emissions to air (group VOC)]	Mass	8,44E-07	kg
Nitrogen oxides [Inorganic emissions to air]	Mass	2,85E-07	kg
Nitrous oxide (laughing gas) [Inorganic emissions to air]	Mass	6,57E-08	kg
NMVOC (unspecified) [Group NMVOC to air]	Mass	3,80E-06	kg
Sulphur dioxide [Inorganic emissions to air]	Mass	4,13E-07	kg

HVO production of Palm and rapeseed oil (Tivander, 2013)

Inputs			
CH: natural gas, low pressure, at consumer [fuels]	Energy (net calorific value)	4880	MJ
CH: transport, lorry 28t [Street]	Ecoinvent quantity ton kilometer (tkm)	246	tkm
CH: transport, lorry 40t [Street]	Ecoinvent quantity ton kilometer (tkm)	200	tkm
OCE: transport, transoceanic tanker [Water]	Ecoinvent quantity ton kilometer (tkm)	23886,6	tkm
RER: phosphoric acid, industrial grade, 85% in H2O, at plant [inorganics]	Mass	0,882353	kg
RER: sodium hydroxide, 50% in H2O, production mix, at plant [inorganics]	Mass	2,4	kg
UCTE: electricity, low voltage, production UCTE, at grid [production mix]	Energy (net calorific value)	138,5989	MJ
Vegetable oil [!Project specific flow names]	Mass	1230	kg
Outputs			
Biogasoline [!Project specific flow names]	Mass	10	kg
Fuel gas [!Project specific flow names]	Energy (net calorific value)	0,32	MJ
HVO (hydrotreated vegetable oil) (kg) [!Project specific flow names]	Mass	1000	kg
Carbon dioxide [Inorganic emissions to air]	Mass	242	kg
Carbon monoxide [Inorganic emissions to air]	Mass	0,0824	kg
Hydrocarbons (unspecified) [Organic emissions to air (group VOC)]	Mass	0,0653	kg
Methane [Organic emissions to air (group VOC)]	Mass	2,76	kg
Nitrogen oxides [Inorganic emissions to air]	Mass	0,26	kg
Nitrous oxide (laughing gas) [Inorganic emissions to air]	Mass	0,0016	kg
Sulphur dioxide [Inorganic emissions to air]	Mass	0,026	kg

Pre and hydro treatment of animal fat and waste				
Inputs				
Catalyst [Operating materials]	Mass	0,68	kg	Ecoinvent Process
CH: tallow, at plant [animal production]	Mass	1230	kg	Ecoinvent Process
Electricity [Electric power]	Energy (net calorific value)	158,76	MJ	PE Process
Natural gas [Natural gas, at production]	Mass	96,75366	kg	PE Process
Nitrogen liquid [Inorganic intermediate products]	Mass	0,28	kg	Ecoinvent Process
Phosphoric acid [Inorganic intermediate products]	Mass	0,75	kg	PE Process
Outputs				
HVO from Waste	mass	1000	kg	Project specific

Assessed emissions per MJ Fuel								
	BR-Ethanol Sugarcane	Other-Ethano	EU-Ethanol	SE-Ethano	EU-HVO-rapese	EU-HVO-anima	EU-HVO-Wast	EU-HVO-Palm oil
CML2001 - Apr. 2013, Abiotic Depletion (ADP el	8,80517E-08	5,85541E-08	2,653E-07	1,25E-09	2,00259E-07	1,30626E-08	4,20264E-10	1,15507E-07
CML2001 - Apr. 2013, Abiotic Depletion (ADP fo	0,156493738	0,125048456	0,5856686	0,036913	0,622863736	0,419269885	0,139981296	0,235084167
CML2001 - Apr. 2013, Acidification Potential (AI	0,000226649	0,000179071	0,0004046	0,000237	0,000740435	3,41518E-05	1,20351E-05	0,000302359
CML2001 - Apr. 2013, Eutrophication Potential (	6,79392E-05	4,77528E-05	0,000492	3,64E-05	0,000661083	1,23254E-05	2,34074E-06	0,000345952
CML2001 - Apr. 2013, Freshwater Aquatic Ecoto	0,010762249	0,006162141	0,0496906	0,000108	0,435880866	0,00113258	5,83531E-05	5,210677171
CML2001 - Apr. 2013, Global Warming Potential	-0,070613171	0,027488543	0,0088643	0,066176	0,105106097	0,021424447	0,003287187	0,072978209
CML2001 - Apr. 2013, Global Warming Potential	0,01590408	0,011441553	0,0771711	0,027833	0,093678608	0,021398866	0,003303702	0,073009421
CML2001 - Apr. 2013, Human Toxicity Potential	0,122479664	0,067178851	0,0391555	0,001962	0,038951994	0,004183734	0,000117376	0,066755899
CML2001 - Apr. 2013, Marine Aquatic Ecotoxicit	7,830645679	5,64931271	27,806574	22,21543	27,73957382	3,238446725	0,243906601	13,52264603
CML2001 - Apr. 2013, Ozone Layer Depletion Po	1,68295E-09	1,19611E-09	5,921E-09	1,1E-10	6,23661E-09	2,6426E-09	2,27684E-12	2,21728E-09
CML2001 - Apr. 2013, Photochem. Ozone Creati	0,000352772	0,000180575	1,926E-05	1,6E-05	1,60185E-05	3,72108E-06	-1,56032E-06	7,7361E-06
CML2001 - Apr. 2013, Terrestric Ecotoxicity Pote	0,004869008	0,002565329	0,0208208	8,94E-05	0,010646973	7,57898E-05	1,17321E-05	0,064514316
	SE-HVO-Tall oil	EU-RME	SE-RME	EU-sugarb	Other-sugarbe	US-corn	Other-corn	SE-biogas
CML2001 - Apr. 2013, Abiotic Depletion (ADP el	8,40464E-08	1,22126E-07	1,221E-07	4,86E-08	3,41E-08	1,33E-07	8,74E-08	6,55E-09
CML2001 - Apr. 2013, Abiotic Depletion (ADP fo	0,282508928	0,409004684	0,4089952	2,53E-01	2,18E-01	1,17E+00	9,84E-01	8,35E-02
CML2001 - Apr. 2013, Acidification Potential (AI	0,000218221	0,000566835	0,0005668	1,89E-04	1,20E-04	5,53E-04	3,76E-04	1,22E-04
CML2001 - Apr. 2013, Eutrophication Potential (	9,07902E-05	0,000425623	0,0004256	1,29E-04	7,07E-05	5,30E-04	2,86E-04	2,71E-05
CML2001 - Apr. 2013, Freshwater Aquatic Ecoto	0,002246494	0,070109456	0,0701078	7,54E-04	1,11E-03	2,41E-02	1,46E-02	1,16E-03
CML2001 - Apr. 2013, Global Warming Potential	-0,023106591	-0,02012815	-0,020128	-4,03E-02	-4,57E-03	-1,75E-02	3,67E-02	-3,95E-02
CML2001 - Apr. 2013, Global Warming Potential	0,021015132	0,051800134	0,0517989	2,69E-02	1,98E-02	1,05E-01	8,00E-02	1,79E-02
CML2001 - Apr. 2013, Human Toxicity Potential	0,010458669	0,019409508	0,0194091	4,27E-03	3,77E-03	2,78E-02	2,08E-02	2,17E-03
CML2001 - Apr. 2013, Marine Aquatic Ecotoxicit	31,55895623	13,16108896	13,160785	5,44E+00	4,49E+00	3,41E+01	2,83E+01	2,03E+00
CML2001 - Apr. 2013, Ozone Layer Depletion Po	1,74742E-09	5,3678E-09	5,368E-09	2,22E-09	1,86E-09	1,00E-08	8,03E-09	7,90E-10
CML2001 - Apr. 2013, Photochem. Ozone Creati	2,46645E-05	1,36518E-05	1,365E-05	6,11E-06	3,91E-06	2,79E-05	2,07E-05	5,24E-06
CML2001 - Apr. 2013, Terrestric Ecotoxicity Pote	0,000270277	0,030560088	0,0305594	-2,73E-03	-1,34E-03	9,27E-04	5,23E-04	6,18E-05