Deliverable 2

Analysis of the best available scientific ILUC research and scientific evidence, key assumptions and uncertainties influencing the ILUC modelling results

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List of Abbreviations

ALCA	Attributional Life Cycle Assessment
CARB	California Air Resources Board
CET	Constant Elasticity of Transformation
CGE	Computable General Equilibrium
CLCA	Consequential Life Cycle Assessment
DLUC	Direct Land Use Change
EU	European Union
FAO	Food and Agriculture Organization
EPA	United States Environmental Protection Agency
FAPRI	Food and Agriculture Policy Research Institute
FSU	Former Soviet Union
GDP	Gross Domestic Product
GHG	Greenhouse Gas
IAM	Integrated Assessment Model
ICONE	Institute for International trade Negotiations (Brazil)
IFPRI	International Food Policy Research Institute
IIASA	International Institute for Applied Systems Analysis
ILUC	Indirect Land Use Change
IMAGE	Integrated Model to Assess the Global Environment.
IMAGE-LPJmL	Lund-Potsdam-Jena model with Managed Land model include in the IMAGE model
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LCFS	California Low Carbon Fuel Standard
LIIB	Low Indirect Impact Biofuels
LUC	Land Use Change
MPOC	Malaysian Palm Oil Council
OECD	Organisation for Economic Co-operation and Development
NREAP	National Renewable Energy Action Plans (NREAPs)
OSR	Oilseed Rape
PE	Partial Equilibrium
RED	Renewable Energy Directive
REDD	Reducing Emissions from Deforestation and forest Degradation
RSB	Roundtable on Sustainable Biomass
RFS	Renewable Fuel Standard

SOC	Soil Organic Carbon
SRC	Short Rotation Coppice
UNFCCC	United National Framework Convention on Climate Change
USA	United States of America
USDA	United States Department of Agriculture

Executive Summary

According to the European Union's **Directive 2015/1513**, the European Commission has to provide an assessment of the **best available scientific evidence** on indirect land use change (**ILUC**) greenhouse gas (**GHG**) emissions associated with the production of biofuels. The Commission also has to provide the latest available information with regard to **key assumptions influencing the results** from modelling of the ILUC GHG emissions and to assess the possibility of setting out criteria for the identification and certification of **low ILUC-risk biofuels** that are produced in accordance with the EU sustainability criteria.

While Deliverable 1 gives a general overview of available literature and some main conclusions, in this deliverable a more specific **analysis of the ILUC literature** is carried out in order to provide an overview of the state of ILUC science. A **decomposition** method is developed to get a better grasp on the **key assumptions and uncertainties** influencing the ILUC modelling results, while the availability on **empirical evidence** for these components is also provided. Finally, after setting a **selection methodology**, the **most important ILUC studies** are analysed more in-depth.

An **overview of ILUC research** based on Deliverable 1 is presented in chapter 2. 75% of the researchers in the ILUC found is **located** in Europe (of which half in Germany and the Netherlands) and the United States, with the rest mainly in Brazil, Australia and Canada. Forecasts of policy effects is the **goal** of 40% of the studies, and 30% is on preventive measures. Recognition of biofuels potential and analysing the effect of regulation are other goals. More than 40% of the **studies** is economic, followed by 20% LCA studies and 28% deterministic studies. Integral assessment is only in 5% of the studies. With respect to **feedstocks**, the main focus is on first generation biofuels with corn as the most studies feedstock. Second generation biofuels are studied much less, with a focus on forest residues and straw, miscanthus, and short rotation coppice. Many studies ignore **by-products**, **consumption effects** and **yield effects** of biofuel policies.

Chapter 3 develops some **important topics in ILUC research**. It includes first a **decomposition of ILUC** in different components and assembles available information on these components. Second and derived from the lack of knowledge problems in **uncertainty analysis** are discussed. This is followed by a discussion on **strategies to reduce ILUC** after which some comments are made to put the ILUC analysis in a broader perspective.

ILUC values for pathways

Results of recent ILUC studies are **far from consistent in their outcomes**, and after 2012 there seems to be **no further convergence in results**. In the table below we must be aware that the CARB estimates are for the US, implying that they could be different. However, Gohin (2014b) suggested that when GTAP calculates the effect of European rapeseed oil the differences in their results were the consequence of problems in the database and that after correction the US and EU results were almost the same. Therefore, it seems that the table gives an indication of the differences in studies.

Table 1 ILUC comparison per feedstock (gCO_2-eq/MJ) between 5 studies. **Source:** own compilation

	Laborde (2011, p. 78)	Valin(2015)	CARB(2009)	CARB(2015)	GTAP-EU (2013)
wheat	14	34			10
maize	10	14	45	30	7
sugar beet	7	15	0	0	16
sugar cane	13	17	69	18	32
rapeseed oil	54	65	63	22	19
soybean oil	56	150	95	44	28
palm oil	54	231	0	107	24
* US outcomes; Adjusted towards a 20 year period					

The basic conclusion must be that especially with respect to **biodiesel** the change from results around 2010 and results in 2015 have a **different direction in the US compared with the EU**: while Valin (2015) has significantly higher results for soybean and palm oil than Laborde (2011), CARB reduced its estimates for biodiesel emissions.

Furthermore, we have seen that even with **comparable results** between the Laborde (2011) and Valin (2015) study the mechanisms behind these results are fundamentally different with Laborde having mainly land use change in regions far away, and Valin mainly forest reversion emissions and other natural land conversions in the EU.

In interpreting the results above, one should be aware that the results include the effect of reduced consumption, and therefore if one would like to exclude this from the ILUC factors, one should increase the ILUC factors with 30% to 50%.

Emissions from **second generation biofuels** are **not analysed much**, but they tend to be lower than for first generation biofuels, and sometimes negative emissions are generated if it is possible to use low carbon land and increase the carbon stock of this land or the vegetation on it by perennial crops.

ILUC decomposition

The figure below provides an overview of the suggested **decomposition method** for a specific pathway in hectares per TJ. In this case the area feedstock per TJ of biofuel is 20 ha. However, because co-products are produced, these substitute for 1 ha of the feedstock, 2 other ha of other crops, implying 3 ha for all crops together, and also 0.5 ha of grassland, implying that agricultural area growth is 3.5 ha less than the original 20 ha needed to produce the feedstock.

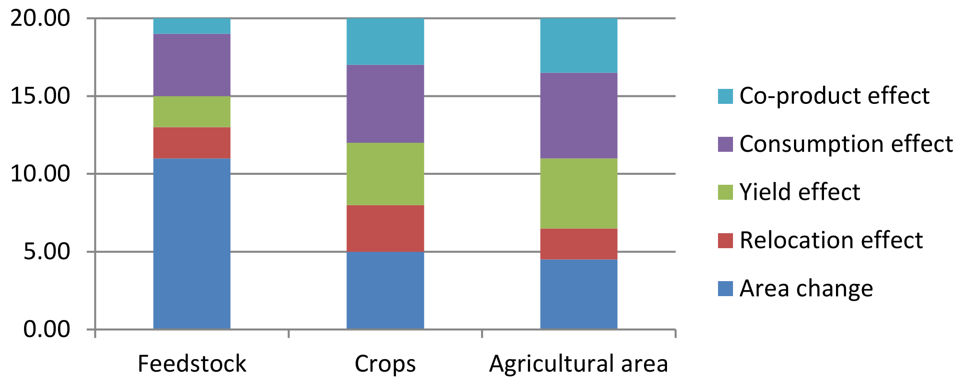

Figure 1 Overview of the decomposition method. **Source:** Own work.

The increase in land requirements is further reduced because the area needed for the biofuel gives pressure on the land market and maybe other input markets and therefore land prices commodity prices rise. The increase in prices induced a reduction of consumption for feed, food, or other non-biofuel uses. Reduced consumption reduces the area expansion. Again, the reduction of consumption not only reduces the area expansion of the feedstock, but may also reduce the area for other crops or livestock. Third, because of the higher commodity and land prices, it may be beneficial to increase yields. The feedstock yields increase, but also the yields of other crops, and maybe also the yields of grassland. Finally, all substitution processes because of price changes imply that different commodities are produced having different area requirements per unit of output, and maybe also that production takes places in different regions with different yields. This is included here as the relocation effect, that can be both negative and positive. What is left over is the total area change. In most cases the increase in feedstock area will be more than the increase in crop area, and this will be more than the increase in agricultural area. When needed, we could add a fourth column that includes agricultural area plus commercial forest.

The decomposition approach above provides the decomposition in hectares. In order to understand the decomposition further, one has to relate the changes in hectares into greenhouse gas emissions. In order to do this one must know which land is converted, i.e. the location of area expansion, and the greenhouse gas changes involved with these changes, i.e. the emission factors. In the following sections we describe the line of reasoning a little bit more in detail.

Key assumptions influencing modelling and its empirical foundation

Yields are an obvious factor determining LUC per MJ biofuel, but less important than you would expect at first sight. Next to the relevance of yields for direct land use for biofuels, **yield projections** in combination with other factors that determine trends in land use have also another consequence that is relevant for GHG emissions from LUC. If in the baseline agricultural area is reduced because less land is needed for the production of agricultural commodities, it may be that carbon consequences of land use change are much less than in the case that agricultural area is expanding into pristine areas already in the baseline.

With respect to **co-product accounting** most approaches start with calculation in weight or energy share, but in more complex models the production of co-products may generate very diverse effects depending on the commodity substituted and the land requirements for this commodity. In for example Laborde (2011) and Valin (2015) co-products production generates for example land expansion for vegetable oils in South East Asia. It is difficult to trace to what extent this is realistic.

The share of LUC reduced through **food, feed** and **other consumption**, both for the feedstock, other crops, and livestock, depends basically on **supply and demand elasticities**. Especially supply elasticities are much larger in the long term than in the short term, and even estimates of short term supply elasticities are very scarce. As a consequence the size of the consumption effect is very uncertain, where it may be argued that it should not be included in ILUC of biofuels because the reduction of ILUC is caused by the reduction in consumption.

The **share of LUC reduced** through **increased yields** because of price increases of the feedstock, other crops, livestock products and land is crucial in many ILUC studies. This share is determined both by the difference between average yields and yields on new land and intensification of current land use. It is basically determined by a combination of land expansion and yield elasticities. While sound econometric estimates suggest very low yield elasticities and larger land supply elasticities, all these estimations generate basically short term yield and area elasticities and most of them are focused on specific products instead of crops or agriculture as a whole. Long term elasticities may be larger than short term elasticities, both for yields and area.

Therefore the yield effect is also very uncertain.

Yields depend on the **region** where production takes place. Therefore, if international trade relocates agricultural production the required area may change. Furthermore, the location of cropland and pasture land expansion may also influence the carbon changes of land conversion. Also substitution between different agricultural commodities with different yields may influence agricultural land use. Different models use different assumptions on trade, but empirical evidence on trade dynamics is relatively poor.

In order to analyse the effect of cropland or agricultural **land expansion**, one first has to know at the cost of which land type expansion takes place. There is not much known about this especially because expansion patterns may change easily over time. And if you know where it takes place, the emission factors of land emissions have to be determined. With respect to uncertainty in carbon accounting: available data on biomass are uncertain, estimates of soil carbon fluxes from land-cover change, where the remote sensing used to allocate land cover are highly uncertain.

Uncertainty

The fundamental point is that empirical evidence on the components of LUC emissions is very meagre. Supply and demand elasticities are uncertain, and this is even more for area versus yield elasticities, and which land is converted. Also the precise substitution process of biofuel co-products in animal feed is very complex. Perhaps the least uncertain is information on the GHG releases per type of land cover change, but also there the spread is large. Because there is fundamental uncertainty, i.e. probability distributions are not known, it is very difficult to do sensible Monte Carla analyses, and one may argue that it is better to analyse the sensitivity to specific parameters by comparing different scenarios with different parameters. However, all the uncertainty analyses accomplished generate wide ranges for ILUC emissions of current biofuel policies.

ILUC mitigation options

Low ILUC biofuels

Section 3.5 investigated strategies to reduce ILUC. The first strategy focuses on low ILUC feedstocks. One type of low ILUC biofuels is to produce them from co-products like straw and forestry residues. There seem to be opportunities from an ILUC perspective, but one has to take into consideration that:

- Harvesting residues may be at the cost of organic soil carbon

- Harvesting residues may provide incentives to switch to techniques with lower productivity for the main products
- Harvesting residues for biofuels may be at the cost of using them for other purposes.
- Harvesting residues for biofuels must also be cost effective

Use of marginal land

The second strategy is to grow feedstocks on marginal lands, i.e. land that is not used for other purposes. When perennials are used on degraded or low carbon land that would not be used otherwise, the carbon value of the biofuel feedstock may be higher than the carbon value or carbon sequestration potential in the original vegetation, generating negative emissions from land use change. However, be also aware that the marginal land could also have been used for the production of other commodities like paper pulp that may reduce production of these commodities elsewhere reducing the pressure on pristine areas or releasing agricultural land.

Yield increases

A third strategy is to increase yields. Several studies suggest that investment in R&D and extension services has high returns. However, if you require these investments to certify biofuel production, it is basically a conditional sale. So, if these policies are useful for biofuel production, why wouldn't you apply them also to food production?

Protecting high carbon stock areas

A fourth strategy is the protection of areas with high carbon stocks. An important aspect is that policies to avoid conversion of natural vegetation are not necessarily the result of the use of biofuels or policies that stimulate the use of biofuels. In other words, the benefits of protection of natural vegetation and lower ILUC emissions from food and biofuels production cannot be allocated to the production of biofuels only, unless these policies are implemented as part of the policies that stimulate the sustainable production and use of biofuels. Moreover, the protection of natural vegetation may limit the ILUC emissions of biofuels, but this may also lead to a trade-off with higher food prices and higher impact on food consumption.

Certification

In general it can be concluded that the certification of low ILUC and ILUC free biofuels is unlikely to be able to avoid all indirect effects. Additional measures, beyond the scope of certification, are therefore needed, such as integrated land use planning including territorial policies.

Impact of non-biofuel policies on ILUC

Some non-biofuel policies may influence the ILUC effect of biofuel policies. It may be worthwhile to be aware of this. We discuss shortly the effect of agricultural, environmental and climate, trade and R&D policies,

Agricultural policies

Some agricultural policies may have consequences for LUC of biofuels. First, farmers get a CAP premium if they keep their land in good agricultural and environmental conditions, even if it is not or marginally used for production. This implies that it will be ploughed preventing carbon sequestration. Second, if agricultural policies promote less intensive schemes with lower yields, then this may reduce the amount of land that can be used for biofuels. Also for example animal welfare regulation, set-aside land policies, and tillage requirements for CAP subsidies may influence the type and amount of land that will be converted. Third, subsidy policy is an important aspect of agricultural dynamics. Decoupling of subsidies reduced prices and made European feed more competitive with imported feed. Fourth, also rural development policy is sometimes focused on improving yields, improving infrastructure which may provide the same type of effects as R&D and extension policies. A last example may be the effect of for example the abolishment of sugar quota in the EU that may increase the opportunities to increase sugar beet use for biofuels that has according to some studies relatively low biofuel consequences.

Environmental and climate policies

Environmental policies may reduce the opportunities to convert high biodiversity land which may also restrict possibilities to expand in high carbon land as a consequence of biofuels policies. On the other hand, if environmental legislation in the EU restricts possibilities for land conversion, it may also be that instead of this land outside the EU is converted with much higher GHG emissions.

A consistent climate policy that also prices land conversion and GHG sequestration of forests, may reduce ILUC a lot. For example, Valin et al. (2015, p. 39) calculate that a price of 50$ per ton CO_2 would reduce LUC emissions from the EU biofuels policy from 97 g CO_2/MJ to 48 g CO_2/MJ, and if peatland would not be allowed to be converted to 4 g CO_2/MJ. Policies like REDD+ to prevent forest conversion are meant to accomplish some pricing of carbon in forests.

Trade policies

More flexibility to import biofuels potentially provides an opportunity to reduce GHG emissions. For example, the direct emissions from sugar cane ethanol are much lower than for maize or wheat ethanol, although the indirect emissions depend a lot on land use policy in the producing countries.

More flexibility in trade of crops and livestock in general may change the international relocation of land. If increased biofuels production in the EU is at the cost of other cropland in the EU because other regions are more cost competitive, while these other regions have lower yields or other reasons for larger GHG emissions, the indirect land use effects of EU biofuels may increase with more free trade.

On the other hand, trade policy can also be used as an instrument to force third countries for stricter compliance to environmental regulation. If tariff reduction in free trade agreements are made conditional on environmental policies, then LUC of biofuels may be reduced. However, such conditions have to our knowledge nowhere been implemented.

R&D and extension policies

First, research on 2^{nd} generation technologies and technologies to improve yield on marginal land may result in the development of low LUC biofuel pathways. LUC is roughly proportional to direct land use.

Second, research leading to increasing yields for biofuel feedstock will reduce direct land use change and land requirements for non-biofuel purposes, and therefore also indirect land use change.

Third, research leading to a general increase in yields will free land that is needed for non-biofuel purposes and this land is low carbon land without competitive uses that may be used for biofuel production.

What has been said about R&D holds also for extension policies that are meant to spread the knowledge that has been generated by R&D. R&D without diffusion of the knowledge that has been generated is not effective.

Lessons from in-depth analysis of some ILUC studies

The in-depth analysis of some well-known and policy-relevant ILUC studies shows that within the group of economic models the essential difference is not between partial and general equilibrium, but about the question what **mechanisms are included in the model**. However, although the mechanisms are not fundamentally different, the outcomes are and the main mechanisms behind the end results are. For example, if you compare the land use changes of the recent CARB estimates for corn ethanol with those of Valin (2015) and Laborde (2011), Valin has a cropland use change of about 0.9 ha/TJ, Valin of 7.7 ha/TJ and CARB(2015) of about 5 ha/TJ. Despite the difference in land use expansion, the GHG emissions don't differ much between the Valin and Laborde study, while CARB has more or less double the land use emissions from those calculated by Valin despite that the area effect is smaller. In summary, even emission factors that look more or less the same have a completely different dynamics.

Analysing results from different reports is very labour intensive, and in the end it is impossible to derive the main mechanisms from the reports. This is something recognized also by for example Searchinger et al (2015), Tyner et al (2016) and Malins(2014). It is essential to open the black boxes behind the reporting of the model results.

The essence of the results can be explained by a limited number of shares and other parameters, that are related with combinations of model parameters. For example, the consumption effect depends on demand, yield and area elasticities, where the percentage of production increase accommodated by yields depends on the area and yield elasticities together.

This implies that interpretation of the literature should be based on the main explanatory parameters that are distilled out, more or less like in the ICCT study, and the analysis performed in sections 3.2.-3.4 of this report. We have seen in chapter 4 that this is not an easy task requiring further investigation.

A broader perspective

Finally, as a comment outside the direct ILUC emissions analysis, one should take into account **economic attractiveness** of the options. As far as reduction of GHG emissions is the goal of biofuel production, it is important that the cost of reducing these GHG emissions is in line with other opportunities to reduce GHG emissions. Furthermore, if you include ILUC reducing mechanisms as reduction of food consumption in the analysis, one should include **all the GHG consequences** of the biofuels policy into consideration, including N_2O and CH_4 emissions from intensification, rebound effects through lower fossil fuel prices, effects on other substitutions as a consequence of increasing land scarcities like in the forestry, building and chemicals industry. And finally, in implementing a biofuels policy, the **ILUC effect** of biofuels **is only one of the many factors** that should be taken into account in an impact assessment of such policies.

1. Introduction

The European Union (EU) developed a renewable energy policy in order to fulfil its commitment to mitigate greenhouse gas emissions. The Renewable Energy Sources **Directive (RED)** sets a target of 10% renewable energy in transport, the majority coming from biofuels. Although mandatory sustainability criteria require that unsustainable land conversion into high carbon or high biodiversity land is not allowed, this doesn't guarantee that as a consequence of biofuels production unsustainable land use for other purposes is created. If land for biofuels is converted from cropland or grassland the production on those land has to be grown somewhere else, and if there is no regulation that this must happen sustainably, this may happen in an unsustainable manner. This conversion may take place in other countries than where the biofuel is produced. This is called **indirect land use change (ILUC).** In 2015 it has been decided that measures to reduce ILUC will also be included in the RED, although it is only a reporting requirement.

This report will provide inputs for the reporting requirements of the EC by summarizing and interpreting the **best available scientific evidence** on indirect land-use change emissions associated with the production of biofuels (art 3(1) of the ILUC directive), with a special focus on the **key assumptions influencing the results** of modelling indirect land-use change greenhouse gas emissions, including measured **trends in agricultural yields** and **productivity, co-product allocation** and **observed global land-use change** and **deforestation rates**. Related with this **uncertainty** estimates of different biofuel pathways will be evaluated. As far as possible some information will be assembled on the **impact Union policies**, such as environment, climate and agricultural policies on ILUC. It will also analyse the scientific evidence on **measures** (introduced in the directive or not) to limit indirect land-use emissions, either through promotion of low ILUC biofuels or more general measures.

In this report we will **analyse** the scientific LUC research identified in Deliverable 1. First, in chapter 2 we provide an **overview** of the current ILUC research. In chapter 3 we develop a general **decomposition scheme** for LUC (3.2), and summarize **empirical evidence** on the different steps in LUC change analysis (3.3). This will give an insight in the main **conclusions and uncertainties** of research on LUC of first generation biofuels and biofuel policy in general. Attempts to analyse uncertainty in a systematic way will be discussed, with the conclusion that results of Monte Carlo analyses are difficult to interpret, and that some studies therefore do some sensitivity analyses for specific parameters.

Next to analysing outcomes and uncertainties of first generation biofuels under current circumstances, one may also investigate to which extent biofuel pathways or different policy options exist to reduce ILUC. A broad category of **mitigation options** is to improve environmental regulation in the countries with carbon intensive land use changes. Another options is to search for low ILUC biofuel pathways. Both are discussed in section 3.5.

After having discussed the general issues of ILUC, in chapter 4 some recent studies are investigated more **in-depth**. Three **representative studies for the EU**, of which one before 2012 because it is an important point of reference, are included, as is the other main approach used by the California Air Resource Board in the US. Two studies are discussed to get a better grasp on decomposition and available empirical information. The focus is on explaining the results of the models by implicit or explicit assumptions on the different components of ILUC as described in section 3.2. It becomes clear that it is not easy to get information on all components out of the reports, and that different reports have different mechanisms included.

Finally, we **summarize** the results and policy options in chapter 8.

2. Overview of scientific ILUC research

In this section an **overview of the scientific ILUC research** is described, which is carried out as part of the analyses carried out with respect to 'Literature review and systematic overview of results of ILUC research and the available scientific evidence' (Deliverable 1) and the 'Analysis of the best available scientific ILUC research and scientific evidence, key assumptions and uncertainties influencing the ILUC modelling results' (this report, Deliverable 2).

The initial literature search returned 1248 entries. This literature was narrowed down through **a 1st preselection** which excluded studies focusing on aspects which were not of direct interest to this study, i.e. studies focusing on biodiversity, water, air quality, (indirect) land use changes from drivers other than biofuels/bioenergy. Furthermore, the 1st preselection divided the eligible literature between studies containing detailed quantitative information and studies which didn't. This was done in order to aid data gathering for Task 2 of the project. After this 1st preselection, there were 168 documents with detailed quantitative information and 328 other eligible documents. All the literature identified in this 1st preselection is included in the database.

A **second preselection** was conducted in order to limit the number of studies to those which would help identifying causes, effects, determinants and mitigation of ILUC for biofuel/bioenergy production. This was done to exclude studies that, even though relevant for ILUC science, did not provide enough information in order to fill in the matrix,. This 2nd preselection yielded 82 quantitative, 123 non-quantitative eligible studies as well as 28 pre-2012 landmark studies. Additionally, a further 49 studies were flagged as possibly eligible.

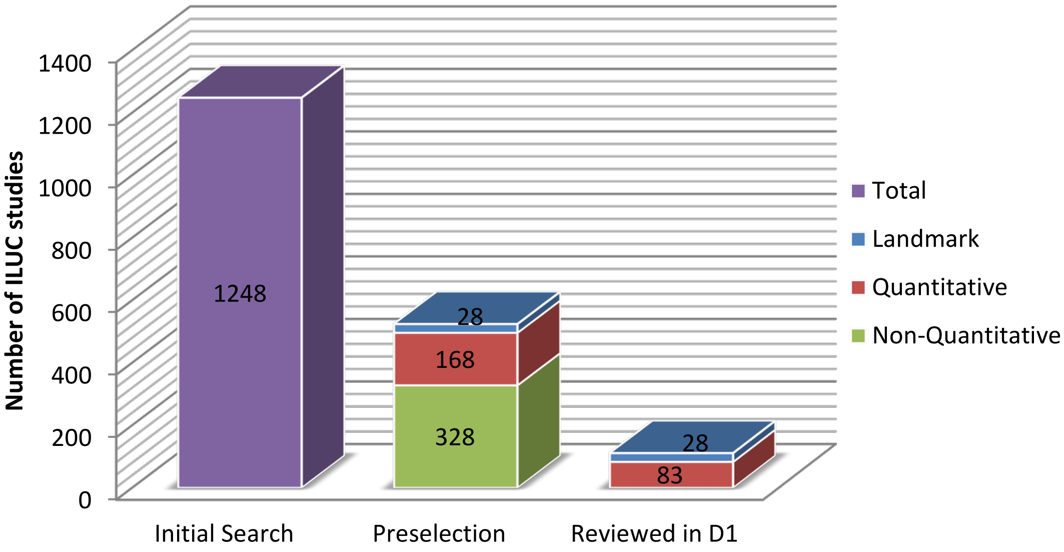

Figure 2 ILUC Scientific Research Selection Process. **Source:** Own elaboration.

Therefore, from the initial 1248 ILUC related studies, 42% went through the preselection and finally 10% were reviewed in D1.

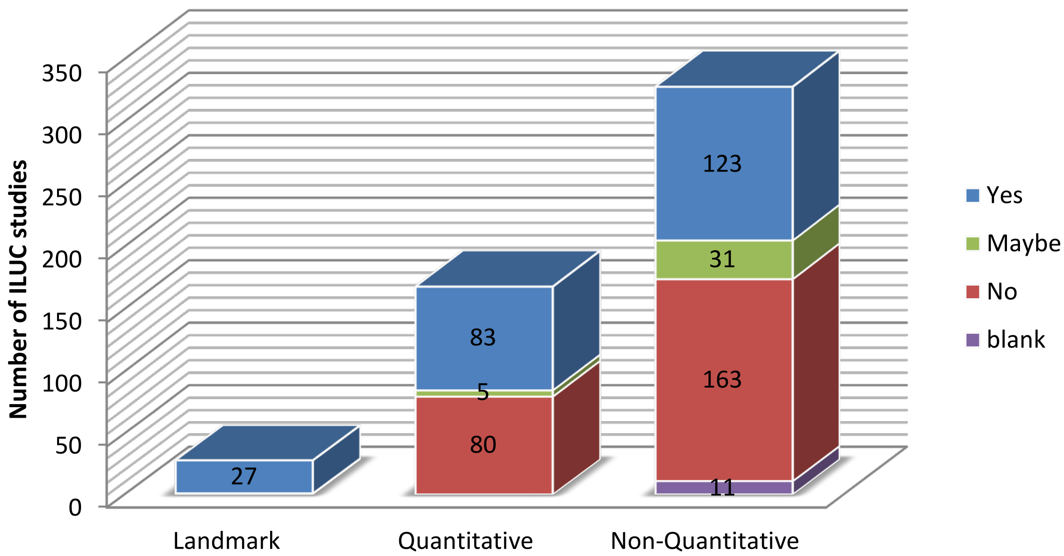

Figure 3 Relevancy of ILUC Scientific Research **Source:** Own elaboration.

In relation to the **year of publication**, 93% of the ILUC related research studies considered have been published after 2012 period, only 7% of the ILUC studies reviewed were published prior 2012. The main reason is that the main objective of this report is to provide a systematic analysis of the latest available scientific research and the latest available scientific evidence on ILUC GHG emissions associated with production of biofuels published in the period 2012-2016 and including the main landmark studies on ILUC published before 2012. Landmark studies were identified as the most cited relevant literature which search engines (Scopus, world of Science, Google Scholar) returned according to the search terms used.

During post 2012 period, 2013 and 2014 were especially active ILUC research publications, while it seems that in 2015 and 2016 the tendency is to be decreasing.

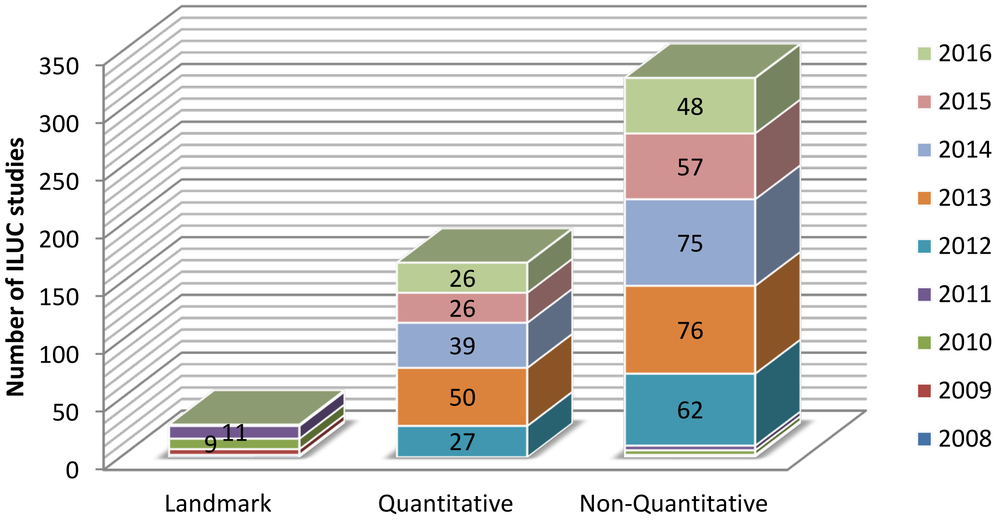

Figure 4 Year of publication of ILUC Scientific Research. **Source:** Own elaboration.

Important fact in the whole overview of available ILUC research results and scientific evidence is the **location** of where worldwide is the research conducted. From authors of the selected studies, it can be concluded that ILUC science is clearly located in **Europe,** which accumulates more than a half of all researchers, followed by the **United States**, which accumulates one quarter of the researchers. The remaining quarter is mainly located in Brazil, Australia, Canada and other countries such as Argentina, Malaysia and Indonesia.

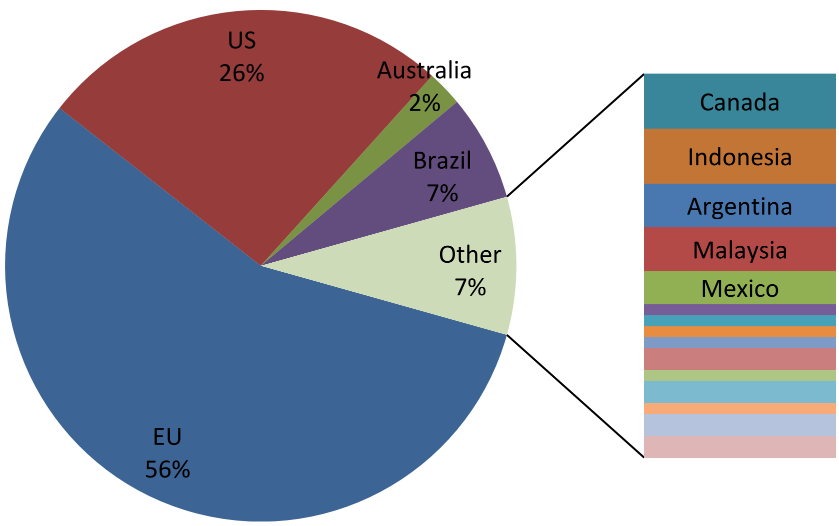

Figure 5 ILUC Scientific Research Location Worldwide. **Source:** Own elaboration.

In Europe **Netherlands and Germany** do clearly accumulate most of ILUC research, accumulating both of them nearly half of the ILUC research conducted in Europe. To be considered also in the picture the U**nited Kingdom, Austria, Belgium, France and Spain** which accumulate together a quarter. The last quarter is finally accumulated by a wide range of countries such as Denmark, Italy, Sweden, Switzerland, Norway, Ukraine and Finland.

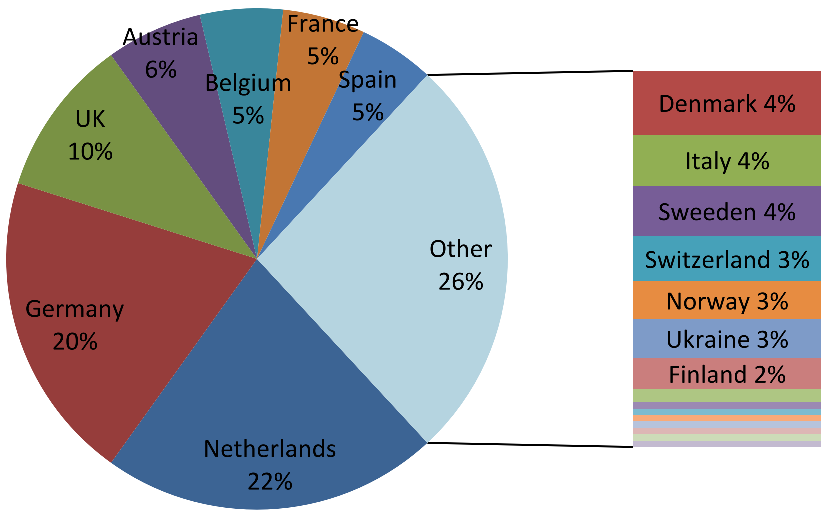

Figure 6 ILUC Scientific Research Location in Europe. **Source:** Own elaboration.

Among the main purposes of the available ILUC research studied in D1, 41% of the papers aim at addressing **Policy Impact Forecast**, followed by **Preventive or Mitigation Measures** which represent 28%. The next purpose of the ILUC research studies is focused on Identification of Biofuel Potential (18%), while Regulatory issues are the least addressed (11%).

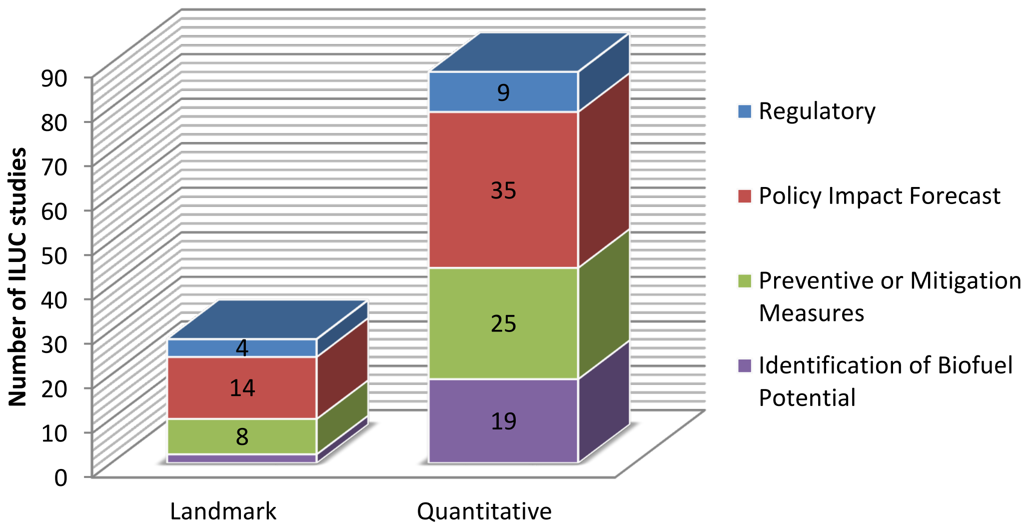

Figure 7 Main Purpose of ILUC Scientific Research. **Source:** Own elaboration.

From the latest available ILUC research, 36% of the studies are **Model Projections (including LCA)** and 24% **Landmark Studies**. Review Studies, Case Studies and Discussion or Methodological Studies follow these in a very similar proportion that ranges around 12-16%.

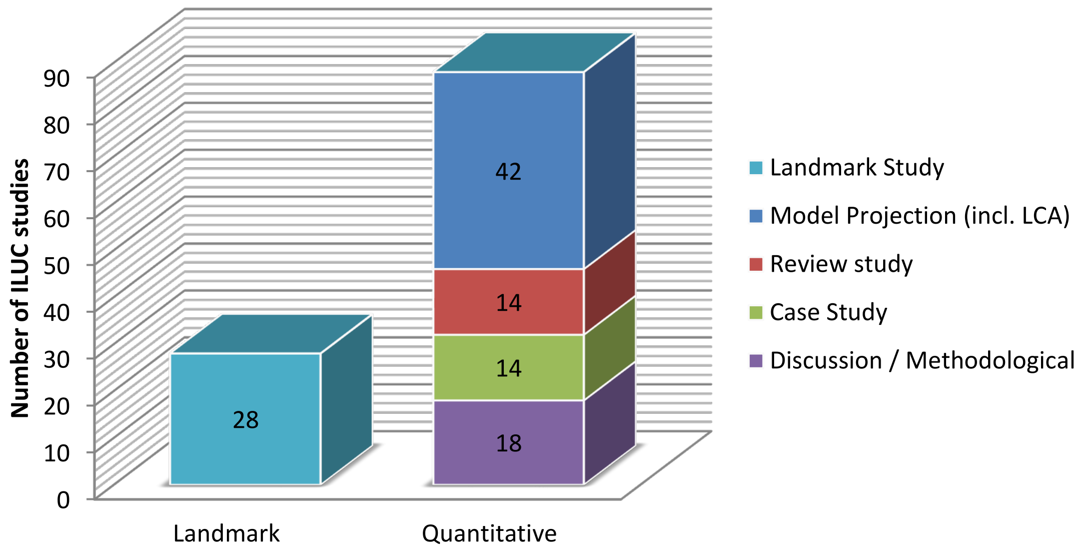

Figure 8 Classification of ILUC Scientific Research. **Source:** Own elaboration.

Following a **comparative analysis of different approaches and methodologies** to evaluate LUC GHG emissions of biofuels is presented. For each of the methodologies used nowadays, the main **rationale and scientific evidences** behind are compared, followed by the **uncertainties** and **sensitivities** that these present. Finally the main **models** used in each approach, as well as the **geographic scope** are presented. Finally the **range of GHG ILUC results** modelled in each of these approaches are presented.

Table 2 Comparative analysis of different approaches and methodologies to evaluate LUC GHG emissions of biofuels. **Source:** Own elaboration

Methodology	Scientific Evidence	Main sources of uncertainties	Main sensitivities	Models	Geographic Scope	Ranges of ILUC results (gCO_2/MJ). Including outliers	References
Life Cycle Assessment (LCA)	Contains detailed information on techno-economic parameterisation. Poor understanding of land use change dynamics.	Typically LCAs ignore indirect effects. Some studies overcome this by combining them with economic modelling (Hybrid LCA).	- Results sensitive to allocation of ILUC to all products of a given process, or to biofuel only. - Technological setups and feedstock possibilities.	LCAs and Hybrid LCA	Multiple, depending on study. Always local.	Biodiesel: 9 – 79 1st Gen.: 4-72	(Acquaye et al. 2012; Acquaye et al. 2011; Bento & Klotz 2014; Boldrin & Astrup 2015; Fargione et al. 2010; Prapaspongsa & Gheewala 2016)
Partial Equilibrium (PE) Models	Based on the concept of "economic equilibrium", i.e. supply and demand are equilibrated through price adjustments. Econometric analysis dictates this behaviour.	The models tend to take a regional or global perspective and suffer from uncertainties arising from aggregation: - Crop yields, particularly marginal crop yields. Indirect effects on food consumption. -Broader indirect effects on the overall economy. Especially food consumption - land use change emission factors.	- Feedstock type. i.e. use of maize leads to higher ILUC effects, compared to other crops.	-CARD-GreenAgSim - FAPRI - FAPRI-CARD - GLOBIOM	Regional or global. Mostly covering the EU and US.	Biodiesel: 23 – 231 1st Gen.: 14 – 104 Advanced: -29 – 17	(Dumortier et al. 2011; Edwards et al. 2010; Mosnier et al. 2013; Searchinger et al. 2008; Valin et al. 2015)

General Equilibrium (CGE) Models	Similar to PEs but accounting for the entire economy. Thus include further economic feedbacks ignored by PEs. These are based on input-output tables (i.e. social accounting matrices) with flows usually measured in monetary terms.	Similar to PEs, except that since CGEs include the broader economy: - Their characterisation of agricultural and energy systems is even more aggregate. - Substitution based on elasticities (CET). Parameterisation very uncertain. - Land constraints and land aggregation methods.	- Parametric uncertainty shows that 90% of results are ±20 gCO_2-eq/MJ from the mean (within a single study).	- MIRAGE - IFPRI MIRAGE - LEITAP - GTAP - GREET-GTAP-BIO-ADV	Regional or global. Mostly covering the EU and US.	Biodiesel: 7 – 252 1st Gen.: 1 - 79	(Al-Riffai et al. 2010; Edwards et al. 2010; Laborde 2011; Laborde et al. 2014; Moreira et al. 2014; Plevin et al. 2015a; Tyner et al. 2010)
Consequential/Simulation models	Extrapolations of observed trends and assumptions of future trade patterns, displacement ratios and incremental land use. These methods were developed in order to simplify data intense and complex economic models.	Key assumption is that current patterns are an adequate proxy for potential future ILUC. Thus they do not account for economic feedbacks which may arise.	Unclear due to limited number of studies.	- Original methods	- EU, Canada, Ukraine - EU, US, Brazil, Argentina, Indonesia	Biodiesel: 18 – 101 1st Gen.: 21 – 67 Advanced: 38 - 75	(Baral & Malins 2016; Fritsche et al. 2010)
Historical Approach/ Case Study	Based on case studies and interpreting historical observations.	Counterfactual if biofuels had not been produced. Assumptions are usually based on past behaviour.	Extremely sensitive on assumptions about reduced allocation rules of ILUC factors (similar to LCAs), as well as changes in behaviour, particularly changes in cattle stocking rates and reduced meat consumption.	- IMAGE - In field measurements	Case studies focused in Brazil, Malawi and Germany. IMAGE used in a global study	Biodiesel: 30 – 257 1st Gen.: 1 – 154 Advanced: 1 - 4	(Dunkelberg 2014; Overmars et al. 2011; Overmars et al. 2015)

Regarding type of **modelling** used in ILUC research studies**, Economic Modelling** is the most widespread, accounting 43% of all modelling studies. Among Economic Modelling, both General Equilibrium and Partial Equilibrium Models are used in quite similar proportion of 20% each. Following to Economic Modelling, **Deterministic Approach** and **LCA** account 28% and 20% of the modelling studies respectively.

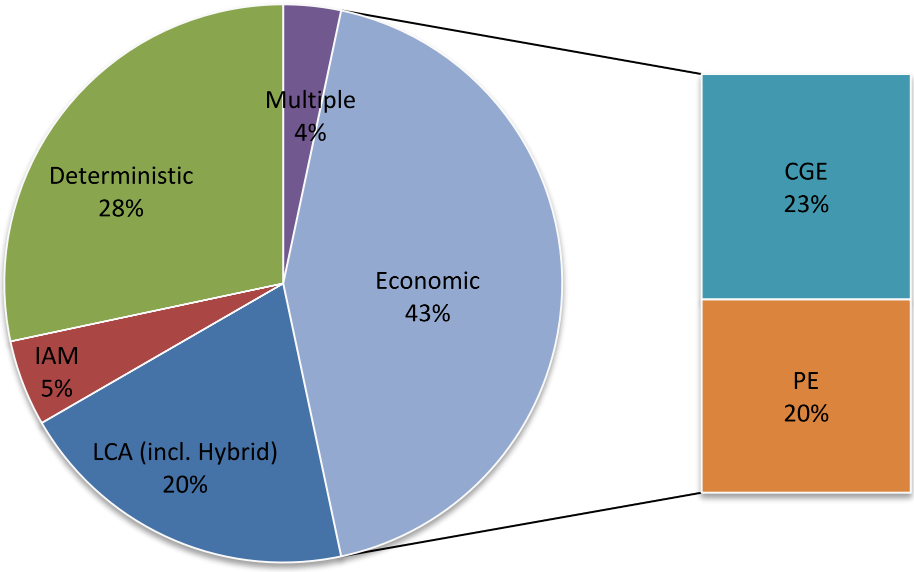

Figure 9 Type of Modelling used ILUC Scientific Research. **Source:** Own elaboration.

It is clear that most of the latest ILUC related scientific researches are **Peer-Reviewed papers**, accounting nearly 90% of the overall ILUC research scientific literature reviewed.

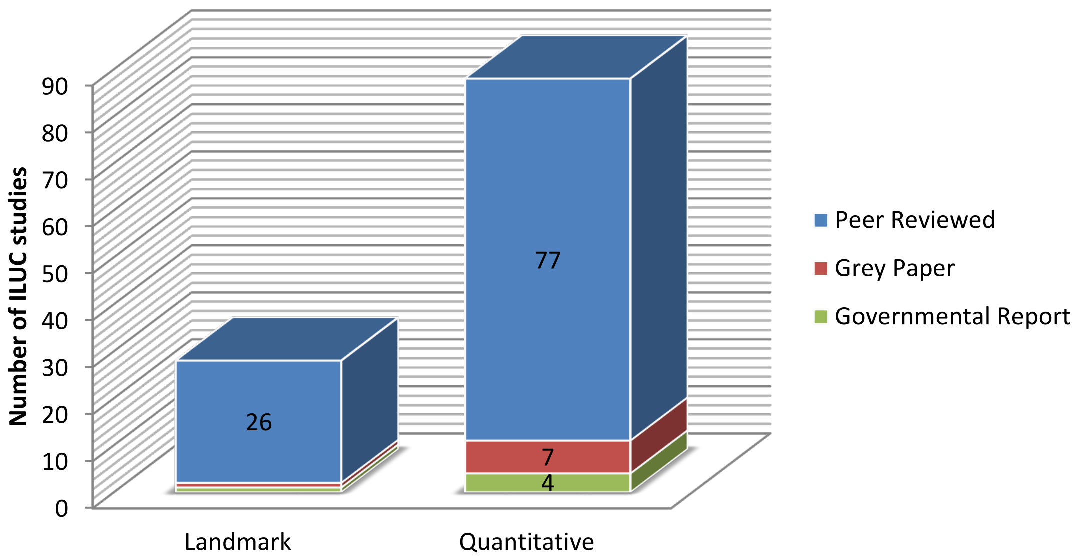

Figure 10 Type of Studies in ILUC Scientific Research. **Source:** Own elaboration.

The **Policy Targets** most commonly addressed in the most recent ILUC research are those from **RED** from EU and **RFS** from US, accounting 9% and 10% respectively. Also to be considered the research addressing **Global Targets** (7%), which is not under any specific Policy Target, but under a Global worldwide Target. However, a very important part of the scientific ILUC research studies (68%) does not indicate this matter. Therefore, it is very uncertain to extract conclusions on this topic.

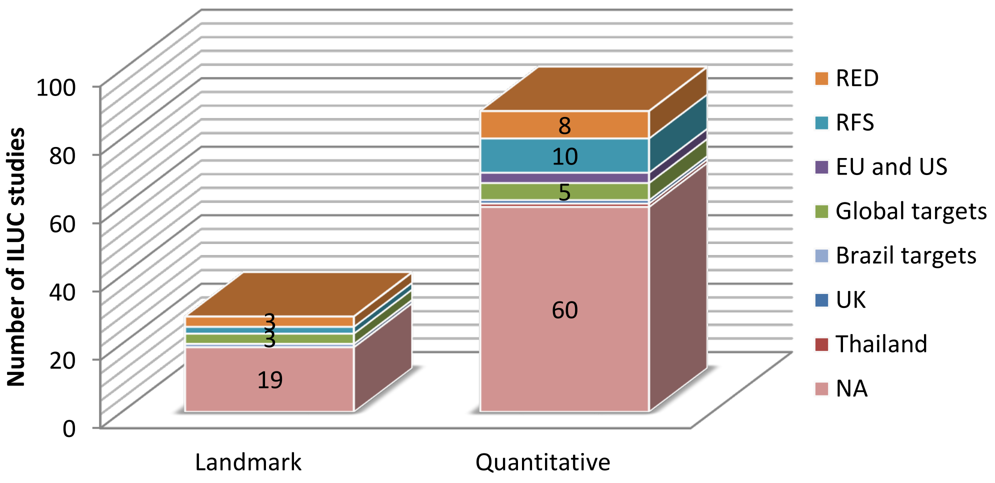

Figure 11 Policy Targets assessed in ILUC Scientific Research. **Source:** Own elaboration.

Regarding the **type of biofuels** most commonly studied in the most recent ILUC research, those are focused in **1st Generation Biofuels** (38%), or cover **both 1st and 2nd Generation Biofuels** (22%). Only 8% of the studies are focused in 2nd Generation Biofuels, and 2% and 3% in bioelectriciy and all energy uses, respectively. It shall be also considered that a important part of the scientific ILUC research studies (28%) does not indicate this matter.

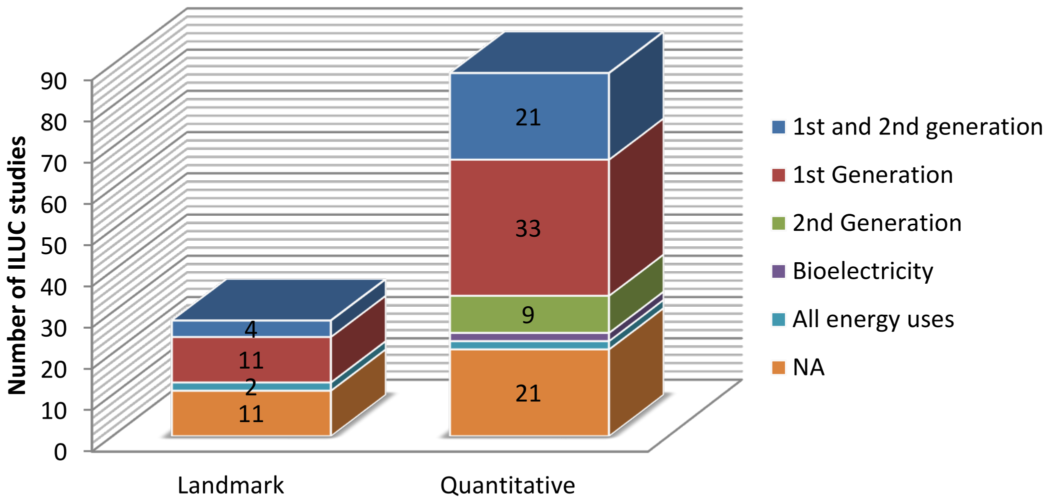

Figure 12 Biofuel Types covered in ILUC Scientific Research. **Source:** Own elaboration.

In relation to the **feedstocks covered** in most recent ILUC research, there is a wide range of feedstock under study. But in detail, more than half cover most important 1st Generation Biofuels Production Crops such as corn (18%), sugarcane (12%), rapessed (11%), soybean (11%), palm (9%) and wheat (7%). Also 2nd Generation Biofuels feedstocks are considered as SRC (5%), forest residues (3%) or Miscanthus (3%).

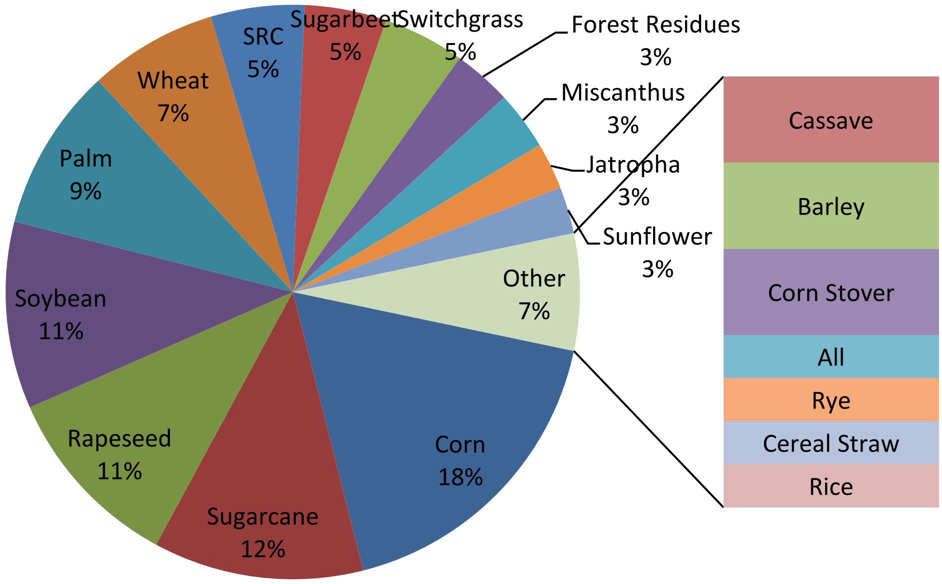

Figure 13 Feedstock covered in ILUC Scientific Research. **Source:** Own elaboration.

Most commonly **demand regions** considered in the most recent ILUC research are EU and US, accounting 16% and 14% respectively. Also to be considered the research addressing Global Demand (14%). However, a very important part of the scientific ILUC research studies (45%) does not indicate this matter. Therefore, it is very uncertain to extract conclusions on this topic.

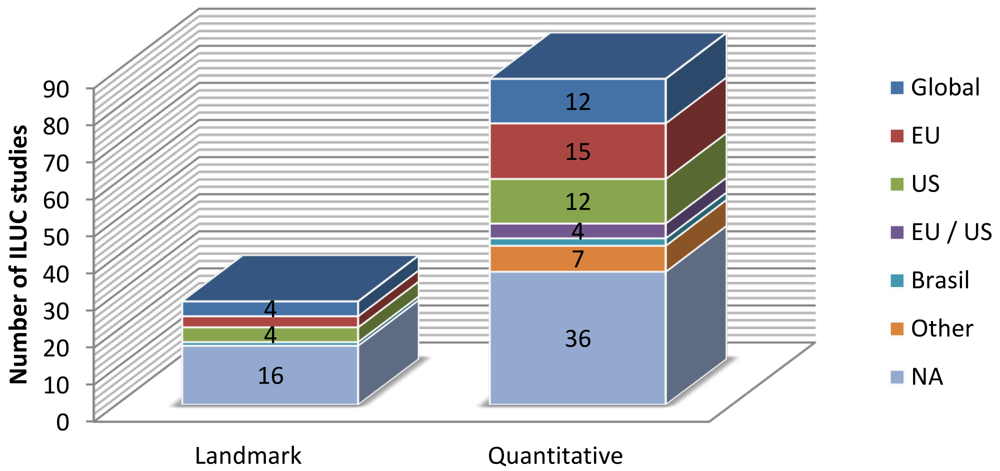

Figure 14 Demand regions covered in ILUC Scientific Research. **Source:** Own elaboration.

On the other side, most commonly **supply regions** considered in ILUC research are global supply (30%), EU (9%), US (13%) and Brazil (7%). However, 35% of the research ILUC research studied does not indicate this matter. Therefore, it is very risky to extract conclusions on this topic.

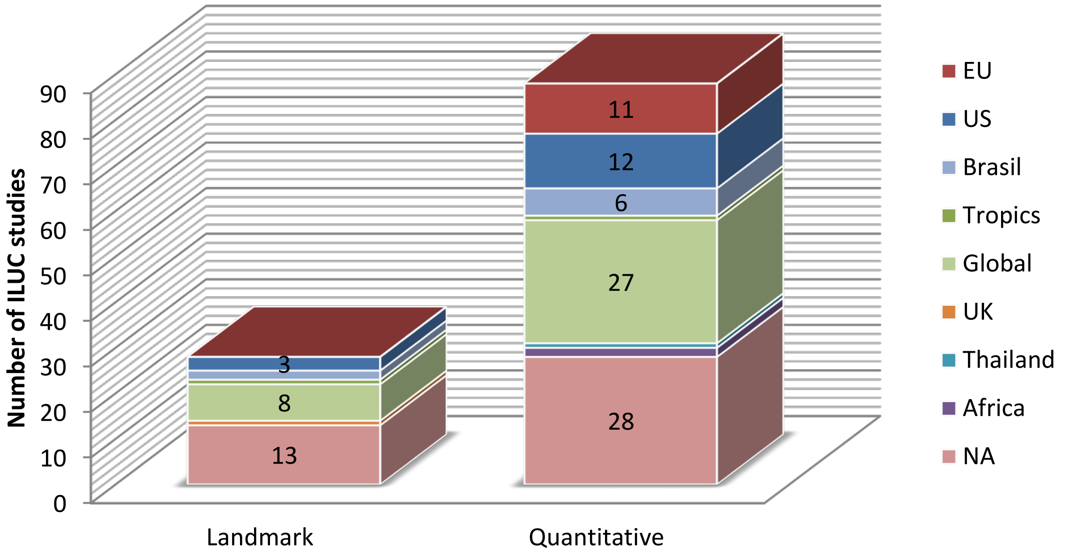

Figure 15 Supply regions covered in ILUC Scientific Research. **Source:** Own elaboration.

14% of most recent ILUC research studied indicates the consideration and **accounting of co-products.** However, 54% of the research does not indicate this matter. Therefore, it is very uncertain to extract conclusions on this topic.

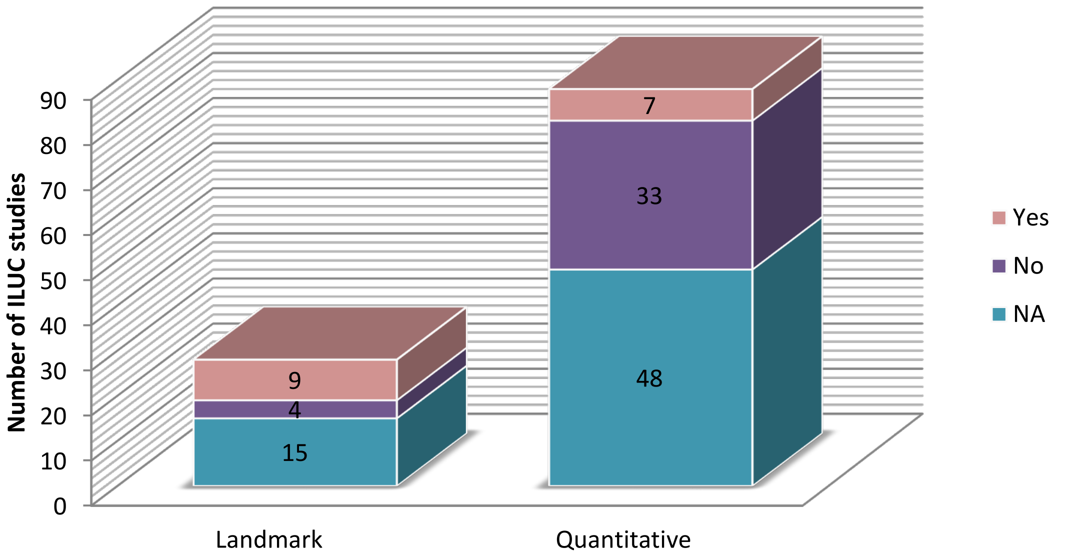

Figure 16 Consideration of co-products in ILUC Scientific Research. **Source:** Own elaboration.

Uncertainty is considered in the 35% of most recent ILUC research studied. It is addressed by different means such as sensitive analysis, or use of different scenarios. However, 51% part of the research does not indicate this matter. Therefore, it is very uncertain to extract conclusions on this topic.

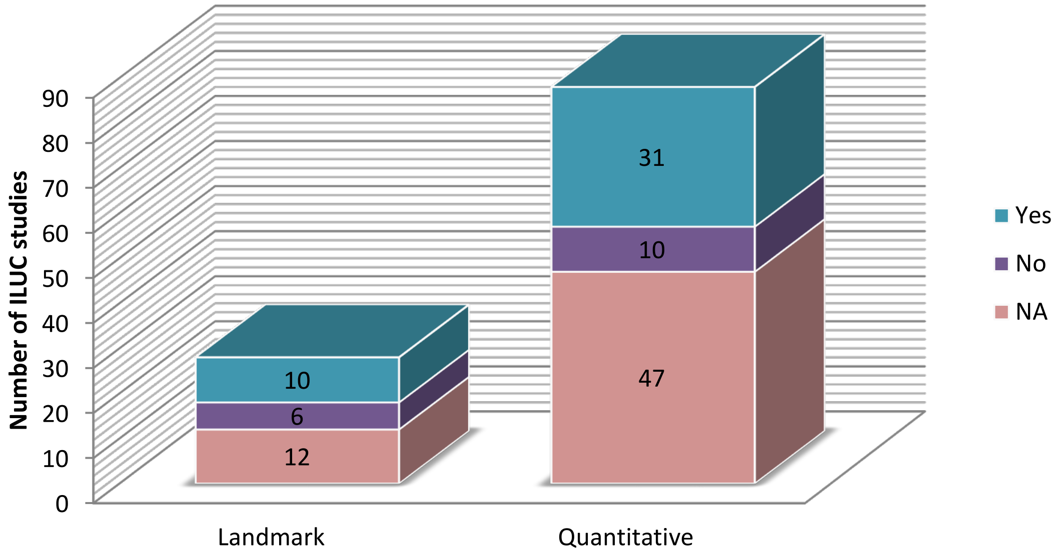

Figure 17 Consideration of uncertainty in ILUC Scientific Research. **Source:** Own elaboration.

Yield increases assumptions are considered in 28% of most recent ILUC research studied. However, 62% of the research does not indicate this matter. Therefore, it is very uncertain to extract conclusions on this topic.

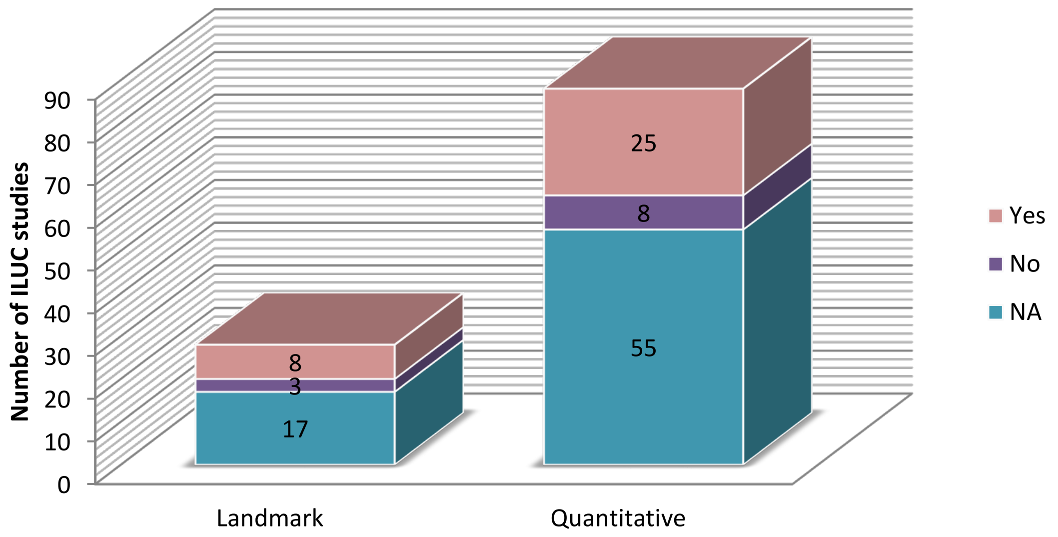

Figure 18 Consideration of yield increase in ILUC Scientific Research. **Source:** Own elaboration.

Based the results of the systematic literature analysis described above and using expert judgement of the project partners a preliminary list of the most recent, influential, detailed, informative analyses is compiled that is considered further more in depth analysis in this report (see Section 6.3 in Deliverable 1). Several more studies are added when and where considered useful and informative for the analysis of the best available scientific ILUC research and scientific evidence presented in this Deliverable 2.

3. Main topics in LUC research

3.1. Latest available scientific evidence

3.1.1. Introduction

The main objective of this report is to provide a **comparative analysis of the latest and best available scientific evidence of the ILUC effects of biofuels**. In the previous section the selection of scientific ILUC research is described and also an overview of the scientific ILUC research is presented. Two important limitations when selecting he best available scientific evidence are related to two conclusions from Deliverable 1:

- The state of scientific knowledge has **not progressed significantly**. There have not been fundamental changes in calculation methods or improved understanding of the involved mechanisms and dynamics. ILUC factors are also fairly consistent across studies (though outliers do exist), especially among studies employing similar calculation methodologies.
- The **uncertainties** related to modelling the ILUC effect of biofuels **remain high**. Factors contributing to this are model parameter uncertainties, differences in crop yields and cropland expansion emission factors. Additionally the definition of different land aggregations affects the potential conversions, real and modelled. This is especially true concerning the possible displacement of "other" oil-crops, which has a large impact on the results.

These conclusions also imply that it is difficult, if not impossible, to determine which of the many assessments can be classified as the 'best available scientific evidence' (e.g. Ceulemans et al., 2012). Moreover, the term evidence seems to suggest that it is assumed that ILUC can be measured. In theory econometric analyses could be used to determine the causal relationship between biofuel production and LUC. However, it is almost impossible to track the consequences of biofuel production changes from other factors that changed during the same period of time, and therefore it is not very plausible that econometric evidence can be found at all.

The selection of the literature considered in this report that captures the best available scientific evidence is therefore inevitably based on a partially subjective judgement of the authors. In Deliverable 1 a list of studies is included to be further analysed in this task. Three recent studies are specifically mentioned, which are considered as the best available scientific evidence. This includes the study of Searchinger et al. (2015), in which various model analyses are compared to show trade-off effects between area expansion, productivity growth and food consumption decrease. The analysis of (Valin et al. 2015) is the most recent study that is focussed on the impacts of EU biofuel use, including ILUC mitigation options and low LUC biofuel pathways. The Laborde assessments (Laborde et al, 2011; Laborde et al. 2014) are the most influential assessments for EU policy analysis, which include recent improvements of the MIRAGE model. In addition, several other assessments are considered based on the criterion that they provide insight in the underlying model assumptions and scientific evidence, particularly with respect to the price-induced yield response, consumption effects and area expansion (ILUC) impacts and the underlying parameters and drivers.

3.1.2. ILUC effects by biofuel type

In Deliverable 1 a systematic overview of results of ILUC research is presented, which includes a **quantitative overview of the ILUC induced GHG emissions by biofuel type**. This overview is based on **28 ILUC studies**. Fifteen studies are based on PE-, CGE- or IAM-models, seven (hybrid) studies use Life Cycle Assessments and four studies are based on historic data. The remaining two studies are based on the ILUC factor approach and on a causal descriptive model.

The resulting ranges of **ILUC emission values per feedstock type** are shown in the figure below. The highest average ILUC factors are found for the production of biodiesel. The mean emission factor is 77 gCO_2-eq/MJ, with palm oil being the worst performing feedstock, and having the highest variation in results. First-generation ethanol has an average ILUC factor of 28 gCO_2-eq/MJ. Only a few studies are found in which ILUC factors of advanced biofuels are calculated, which typically have much lower and sometimes negative ILUC factors, depending on the feedstock and mechanisms that are considered.

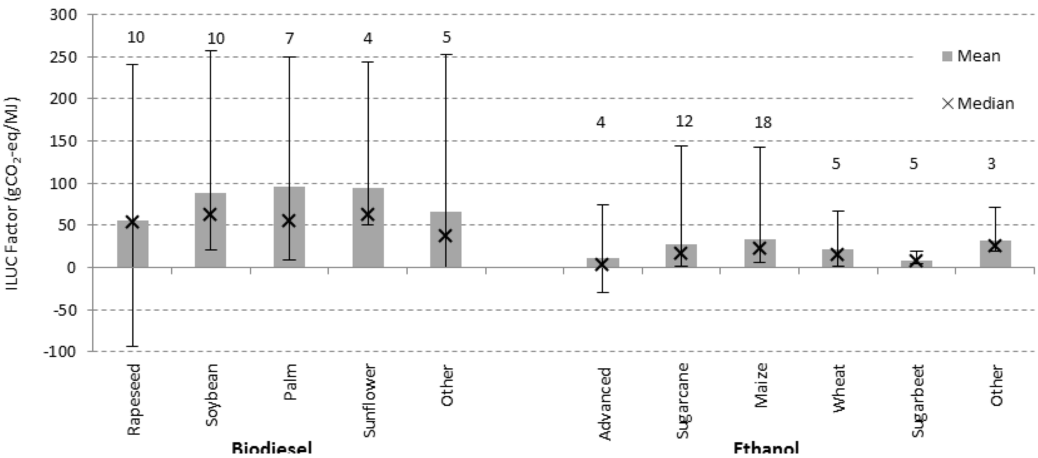

Figure 19 Summary of ILUC factors found in literature for biodiesel and ethanol. Grey bars: Mean, Black crosses: Median, Whiskers: Maximum-Minimum, number of studies calculating ILUC factors written above each column. Note: a given study may include multiple scenarios or feedstocks. **Source:** Own elaboration.

As discussed above, it is not possible to determine which analyses are more scientifically sound or robust and can thus be considered as 'best available scientific evidence'. Therefore, in this deliverable specific attention is paid to differences in approaches and methodologies used and to the underlying variables and the scientific basis of these assumptions.

3.1.3. Impact of RED on ILUC effects

The **RED** requires that the production and use of biofuels meets a set of **sustainability criteria** in order to ensure that their use leads to lower greenhouse gas emissions and does not lead to a loss of biodiversity. Only biofuels that meet these criteria are allowed to count towards the renewable energy targets. The main criteria are:

- A **35% GHG reduction** threshold in comparison to fossil fuels (excl. emissions from ILUC). This target increases to **50%** in 2017 and to **60%** in 2018, but only for new production facilities.
- Biofuels cannot be grown in areas converted from land with previously **high carbon stock** such as wetlands or forests.
- Biofuels cannot be produced from raw materials obtained from land with **high biodiversity** such as primary forests or highly biodiverse grasslands.

The RED also specifically stimulates the use of residues, waste, non-food cellulosic material and lignocellulosic material by allowing that biofuels produced from **residues and waste** count double towards the target (on energy basis) compared to biofuels produced from food crops. Further, the use of food crops for the production of biofuels is capped to 7%, while the total biofuel blend target remains at 10%. Several studies evaluated the impact of the RED criteria on the production and use of biofuels in the EU.

Bottcher et al. (2013) estimated that the global sustainable potential of first generation biofuels is 10 times higher than the RED target in the EU in 2020. In other words, the redirection of sustainably produced feedstock from other applications to biofuels provides high potential compliant with the RED criteria. Based on these studies it can be concluded that sustainability criteria, such as the RED and various biofuels certification systems are only effective in avoiding ILUC effects if **extended to other commodities and countries**. These conclusions suggest that **comprehensive land use planning** and **policies aimed at directly avoiding deforestation and other undesirable land use changes** are more effective compared to sectoral policies due to leakage effects.

These conclusions are confirmed Frank et al. (2012) who use the same GLOBIOM model to demonstrate that the demand for biofuels in the EU can be satisfied sustainability, since most of the global crop production can be classified as sustainable based on the RED criteria. The sustainable potential is more than 10 times the total European biofuel demand in 2020 if the use is reallocated from other applications that do not include sustainability criteria.

It can be concluded that to be effective the RED land use criteria need to be **expanded to other agricultural commodities and regions**. In other words, the emissions from ILUC are best avoided by targeting deforestation and biodiversity loss directly.

Valin et al. (2015) analysed the impact of a maximum share of conventional biofuels from crops on the ILUC emissions, also using GLOBIOM. The cap on first generation biofuels reduces the overall ILUC effects of biofuel use in the EU from 97 gCO2e/MJ to 74 gCO2e/MJ. The reason is the higher share of advanced biofuels with low or negative emissions increases compared to a situation without a cap, so the ILUC effects per unit first generation biofuel is not necessarily lower.

The impact of the cap on first generation biofuels is also analysed by Junker et al. (2015), using the MAGNET CGE model. The results indicate that a limitation of first generation biofuel use in the EU is only partially transmitted to the demand and production of rapeseed and other oilseeds in the EU. More important is a lower import of vegetable oils. Especially vegetable oil production in Argentina and the US are effected, while the impact on the most important vegetable oil producing region (Asia) is limited. The impact on ILUC emissions of the shift in feedstock use is not calculated, but most likely the impact is relatively limited because of the limited impact on palm oil production, which is associated with high ILUC emissions from peatland.

Further, Boldrin & Astrup (2015) argue that legislative frameworks, such as the RED, include calculation methods for calculating the GHG effects of biofuel chains, but there is flexibility with respect to the interpretation of methodological choices and consideration of case specific conditions. Emission savings of biodiesel from rapeseed were determined using five different allocation criteria identified within the calculation methodology described in the Renewable Energy Directive (RED). Depending on the allocation criteria and the system boundary adopted the emission savings range from −34% to 83%.

If the RED GHG reduction criteria would be effective and lead to a reduction of the use of biodiesel from rapeseed, than the feedstock composition of biodiesel production will change which may consequently increase the ILUC effects. Junker et al. (2015) estimated the effects of a ban on the use of rapeseed biodiesel in the EU with MAGNET. This results in vegetable oil from rapeseed being diverted from biodiesel production to food markets. The resulting gap in demand for vegetable oil for biodiesel production is filled by import of palm and soybean oil. These results show that the **RED GHG reduction thresholds might have undesirable ILUC effects**, considering the high LUC effect of palm oil.

Based on the results of the studies cited above it can be concluded that **is unlikely that the RED is able to avoid the impact of biofuel use in the EU on ILUC and biodiversity**. The GHG reduction thresholds are most likely not very effective to control direct emissions of biofuel production and can lead to undesirable ILUC effects.

3.1.4. Conclusion

An overview of outcomes doesn't provide an insight concerning real evidence. Available evidence concerns the parameters, and what is known about them. In the next section we organize the relevant parameters in a decomposition approach and in section 3.3 we describe as precisely as possible what evidence is available, including its status.

3.2. Decomposition approach

In order to get a grasp on the scientific evidence it is essential to understand the **main factors that determine LUC**. Therefore, **decomposition** is essential (Searchinger et al. 2015). However, most studies are very vague in the fundamental components of their LUC calculations. In this section we will develop an overview of the fundamental components of LUC as a consequence of biofuels, where in the next section empirical evidence on these components will be discussed. The decomposition method developed describes the ideal amount of information you would like to have in reports on LUC, but you can find in reporting of results at most a little bit of this decomposition.

The starting point of the analysis is that it should be done **per biofuel pathway**. When the ILUC factors per pathway are known, one can aggregate them to the effects of a biofuels policy, taking into account that there may be non-linearity in the system.

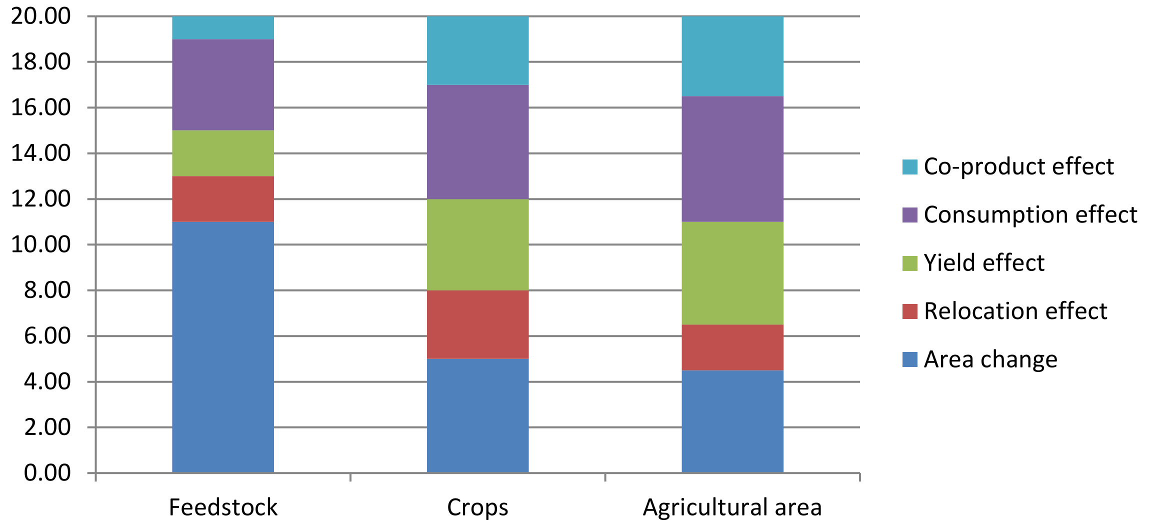

Figure 20 Overview of the decomposition method. **Source:** Own work.

Figure 20 provides an overview of the suggested decomposition method for a specific pathway in hectares per TJ. In this case the area feedstock per TJ of biofuel is 20 ha. However, because co-products are produced, these substitute for 1 ha of the feedstock, 2 other ha of other crops, implying 3 ha for all crops together, and also 0.5 ha of grassland, implying that agricultural area growth is 3.5 ha less than the original 20 ha needed to produce the feedstock.

The increase in land requirements is further reduced because the area needed for the biofuel gives pressure on the land market and maybe other input markets and therefore land prices commodity prices rise. The increase in prices induced a reduction of consumption for feed, food, or other non-biofuel uses. Reduced consumption reduces the area expansion. Again, the reduction of consumption not only reduces the area expansion of the feedstock, but may also reduce the area for other crops or livestock. Third, because of the higher commodity and land prices, it may be beneficial to increase yields. The feedstock yields increase, but also the yields of other crops, and maybe also the yields of grassland. Finally, all substitution processes because of price changes imply that different commodities are produced having different area requirements per unit of output, and maybe also that production takes places in different regions with different yields. This is included here as the relocation effect, that can be both negative and positive. What is left over is the total area change. In most cases the increase in feedstock area will be more than the increase in crop area, and this will be more than the increase in agricultural area. When needed, we could add a fourth column that includes agricultural area plus commercial forest.

The decomposition approach above provides the decomposition in hectares. In order to understand the decomposition further, one has to relate the changes in hectares into greenhouse gas emissions. In order to do this one must know which land is converted, i.e. the location of area expansion, and the greenhouse gas changes involved with these changes, i.e. the emission factors. In the following sections we describe the line of reasoning a little bit more in detail.

Direct land use for biofuels (per pathway)

First, growing of the feedstock of biofuels requires an area that is determined by the energy per kg of feedstock biofuel and the yield of land. This is a straightforward calculation. The result is direct land use expansion for feedstock. The rest of the approach is explaining why agricultural land use change is less than this direct land use change.

Co product accounting

Second, the production of biofuels generates co-products, mainly animal feed, that may reduce land requirements for the production of animal feed elsewhere. This is also a technical relationship, but the calculation about which animal feed is substituted away can be relatively complicated. Furthermore, normally land for other crops than the biofuel feedstock is substituted, so it is more relevant for the analysis of the increase of total crop expansion than for the analysis of feedstock area expansion. A careful analysis of the effect of all substitutions in the chain is required for this, where the reduction in feedstock area is only a part.

Other co-products like electricity have no land use change consequences, but have GHG consequences that normally are tackled in a standard LCA analysis.

Consumption effect

Third, the increase in demand for the biofuel feedstock may generate a price increase of the biofuel feedstock that is also used for other purposes. The increase in price may also generate a reduction in demand for the feedstock for other purposes. This may be because the feedstock is used more efficiently for these purposes (reduction of food waste, for example) or because consumption is reduced. In case of use for food this implies that less is available for nutrition or that waste is reduced. If the lower demand is for animal feed, this will have consequences for other types of feed or consumption of meat.

Yield effect

Fourth, the increase in price may generate a yield increase, reducing the amount of land needed for the biofuel, but at the same time changing land use emissions because of increased fertilizer use and perhaps also emissions related with mechanization. As far as the yield increase is on the area of biofuels, this is included in an LCA analysis, but if it is on crops for other purposes, it is not included in standard LUC GHG analysis, even though it is relevant for the calculation of net greenhouse gasses.

Next to the yield effect there may be an effect on average yields because of expansion of feedstock area when average and marginal yields differ. In most econometric analysis on the yield elasticity the effects are combined, but in most models the difference between marginal and average yields and the increase of average yields on current land are analytically separated. However, the difference is not visible in most model outputs where normally average yields or production and area are reported.

While statistics are normally on harvested area, for land cover the physical area is the relevant criterion. This implies that increase in double cropping or reduction of unused cropland are included in the yield increase component.

Relocation effects

Normally, the exercise above will be accomplished with standard yields, for example the yields in the EU. However, because some production is taking place in other regions with different yields, the LUC changes will be different. Second, there may also be substitution processes going on in animal feeding where area needs per unit of feed differ. These relocation effects have also to be taken into account. They can be both positive and negative.

Net increase in feedstock area

Both the yield and demand effect reduce the expansion of land used for the feedstock of the biofuel. This generates the net expansion of land for the feedstock. One has to be aware that the difference between gross and net land use effect may be at the cost of an increase in GHG emissions because yield increases require for example more fertilizer that generates GHG emissions, or at the cost of food, feed and other non-biofuel consumption in the case of demand reductions.

Analysis at cropland level

This expansion of feedstock area will be at the cost of other types of land. For analytical simplification, we may assume that feedstock production is at the cost of other cropland, and this implies that these other crops have to be produced elsewhere or have to be produced with a higher yield, or consumption of these crops must be reduced. This is the same line of reasoning as with the feedstock area, except for that instead of an increase in demand the driving force is a reduction in area. In both cases the tension must be solved.

However, be aware that the required crop area of other crops may be changed also because of the co-products of the biofuel production, especially animal feed. So, the demand for crops may be reduced as a consequence of substitution of the biofuel co-product. Furthermore, the demand for other crops may be changed because of changes in livestock production caused by price changes of livestock products. All these effects are included in the share of original crop expansion that is absorbed by demand reduction, and these elements may be split out. The end result is the increase in crop area as a consequence of the increase in biofuel demand.

Analysis of pasture land

The increase in crop area will be at the cost of other types of land. For analytical simplification we assume that the cropland expansion is at the cost of livestock area. Again the same line of reasoning can be followed, i.e. the tension between currently used livestock area and demanded area can be solved by yield increases, consumption reduction and area expansion. If it is accomplished through intensification of livestock, this will probably generate extra demand for crops for animal feed, as discussed above, although part of intensification can be accomplished through higher grassland productivity as a consequence of fertilisation, mechanisation or other methods (Lapola et al. 2009). As far as the reduction in land use is caused by reduced livestock production, this may have consequences for meat and milk consumption and food waste reduction. The end result is a net reduction in livestock area or a net increase in agricultural land.

Analysis of commercial forest land

In some studies also consequences for commercial forest land are taken explicitly into account. The expansion of agricultural land gives a pressure on commercial forest land, and again this may be accomplished through consumption reduction, yield increases and area expansion.

Location of area expansion

What is defined as total land expansion depends on the model. In most cases it is cropland expansion, in some cases it is agricultural land expansion and in some cases it is agricultural plus commercial forestry land expansion. When total land use expansion is known, it is important to know which type of land is converted. Depending on the study the end result of the exercise is an expansion in cropland, agricultural land or agricultural and commercial forest land that will be at the cost of the remaining land use types, at least being natural forest land, peat land and other natural land. The distribution of area expansion over different types of pristine areas is the next step in the analysis. This may depend on the region where expansion happens, but depends also on the assumptions on which types of land will be converted in practice.

Emission factors

The end result of the whole exercise is a table of changes of each land use type per region. For each land use change there is a carbon GHG that can be related with it, and the sum of these GHG stock changes are the effects of the land use change as a consequence of the change in land use. Land use change emissions may arise from two mechanisms (i) Loss of carbon stock in above and below ground biomass, (ii) Foregone sequestration (or carbon loss) which would have occurred if the initial land cover remained (for details see Appendix 2, Summary of Biomass Carbon).

Concerning the first, one must be aware that the way in which the land transition is accomplished determines also how much GHG is released. For example, by burning forests all carbon stock will be released in the air, while if wood is harvested the carbon stock will be included in the products made from it, or burned as biofuel. So, the share of carbon stock that is released in the air is also an important factor explaining the emission factors. For the latter (foregone sequestration), assumptions on "counterfactual" land use and climate are important. For instance if biomass production moves onto agricultural land which would otherwise be abandoned, this land may have reverted to natural vegetation, potentially becoming a carbon sink. For both emission types, emission factors are usually calculated over a fixed period (20 or 30 years) and averaged over the years.

Aggregation towards biofuel policies

When one likes to have the outcomes per biofuels pathway, one may aggregate these results towards the totals as a consequence of a biofuels policy. Normally, the outcome of a biofuels policy is just the sum of all the effects of the biofuels pathways used in the production. Analysing how the choice of biofuel pathways is made is normally not part of ILUC studies.

The method in practice

In order to be able to start from the outcomes of the models, one must get information from these models on land use change per land cover type and region.

Also information about changes in production is required as information on changes in animal feeding. The outcome of the exercise is determined by the different shares defined above, behind which there may be different stories.

When analysing what evidence is available, one must analyse what empirical information is available on the different components. That is the challenge of this chapter. In chapter 4 some important ILUC studies will be analysed with the decomposition vision developed in this section in mind. The decomposition developed above is the information you would ideally get from reports on ILUC modelling results. However, reports on LUC results don't report all this information.

3.3. Empirical evidence on key assumptions influencing ILUC

3.3.1. Introduction

There is a large number of simulations and models available, but the essence can be summarized with a limited number of components as discussed in section 3.2. In order to get a better grasp on the fundamental issues we will investigate here the available empirical information on the different components of LUC and what the implications for LUC outcomes are.

3.3.2. Trends in yields and productivity

The first step in the analysis of biofuels is to calculate the **area of feedstock** that is necessary per TJ of biofuel. This area depends on **average crop yields** and **energy productivity** per ton of crop. Especially crop yields develop over time, and therefore are important for area use per TJ. As far as the decomposition method is correct, except for non-linearity that may be in the system, GHG emissions because of LUC are proportional to area per TJ of biofuel.

Global yields for crops increased on average with 2% per year between 1961 and 2006 because of new crop varieties, increased use of pesticides and fertilizers, and improved access to irrigation (Baldos & Hertel 2016; Burney et al. 2010). The **increase in potential yields** is somewhere between 0.6% and 1.1% annually, while closing the gap between potential and actual yields is the other part (Fischer et al, 2014). According to some estimates (Baldos & Hertel 2016) total factor productivity in farms increased only a little bit less than yields.

Future yield growth is difficult to predict. R&D expenditures have been reduced in the 1990s, but increased in the 2000s. Because a larger fraction of current R&D expenditures is private compared with the past, it may be that diffusion of the innovations is more difficult. Climate change may influence productivity growth, where for example increasing temperatures may reduce crop yields while increased CO_2 concentrations in the air may increase yields (Baldos & Hertel 2016). Another issue is to what extent further potential yield improvements are possible. This depends also on the acceptance of new techniques like genetic modification. Roughly, the expectation is that the growth rate of yields will be lower in the future, especially because yields were growing at an arithmetic rate instead of a geometric rate implying that the percentage rate of yield growth is going down (Searchinger et al. 2015).

Future yield growth is relevant to predict the area need of biofuels production, but has in combination with demand factors as population, GDP growth and income distribution consequences for the area needed for non-biofuel purposes and therefore potential land available for biofuel production.

Looking at recent history, Langeveld et al. (2014) show that between 2000 and 2010 global agricultural area was reduced by 47 million ha, partly caused by increased multi-cropping which increased harvested area on the same crop area with 92 million ha (7% of a total of 1.4 billion ha). The background of agricultural area reduction may be urbanisation, tourism and increase of nature area, but also land abandonment because land use is not profitable anymore or because of land degradation. With respect to Brazil agricultural area increased by 12 million ha where 4.9 million ha of harvested area was added by increased double cropping.

Although the main argument of Langeveld et al is that biofuels requiring 14 million ha is a small part of all land use dynamics, one may also conclude that a lot of abandoned and unused land seems to be available that can easily be taken into production without much carbon loss, and especially without deforestation. This, however, will not always happen automatically, because it may be that land has been taken out of production for good economic or social reasons.

Concluding, next to the relevance of yields for direct land use for biofuels, yield projections in combination with other factors that determine trends in land use have also another consequence that is relevant for GHG emissions from LUC. If in the baseline agricultural area is reduced because less land is needed for the production of agricultural commodities, it may be that carbon consequences of land use change are much less than in the case that agricultural area is expanding into pristine areas already in the baseline.

3.3.3. Co-product accounting

The basic idea of **co-products** is that part of the harvest cannot be used for biofuels and is available for other purposes, especially **animal feeding**. In simple models, this is just distributed according to the **weight**, the **energy content** or the **market value**, but in practice **substitution processes** are much more complicated. Examples will be discussed in chapter 4.

Basically, in the more advanced models like MIRAGE and GLOBIOM there is first a substitution between different protein rich feeds and then a substitution between energy and protein feeds. In practice the increase in protein-rich feed by-products leads only partially to a reduction of other protein-rich by-products, while the main effect is that the energy-rich feed is reduced and sometimes also feed from grassland.

Although the substitution process in feed is relatively well described, the real dynamics that is generated is not easy to tackle. The increase in by-products generates a price reduction of protein-rich feed, and depending on the price changes of other feed components total crop-based feed price may either increase or decrease.

In MIRAGE and GLOBIOM, but not in GTAP, an interesting substitution mechanism starts as the consequence of increased protein-rich animal feed. As far as the increase in biofuel by-products is accommodated through a reduction in production of other protein-rich feed, i.e. soy meal, this may generate a reduction of soy production reducing supply of soy oil, which is replaced with the cheaper palm oil that has much less co-products. In this way the positive land use effect of a decrease in land use of feed co-products of biofuels may be compensated by the conversion of peatland and carbon rich forests into palm oil plantations (Malins et al. 2014).

What is the empirical evidence for this mechanism? The basic idea is that recent increases of vegetable oil production are all based on palm oil, implying that an increase of biodiesel also generates an increase in palm oil production. The other idea is more logical in character, assuming that feed rationing is relatively inflexible, i.e. that DDGS or oilcake from biofuels production is substituted by soybean instead of oil cake. It is not clear what the truth is, but it is clear that **models widely differ on assumptions** about this mechanism, generating large differences in LUC GHG emissions from soy and rapeseed oil.

3.3.4. Share accommodated through consumption

Part of the increase in demand for agricultural crops as a consequence of the increase of biofuel demand is accommodated through a **reduction in demand for crops for food, feed and other non-biofuel purposes**. This makes it relevant from an ethical point of view in the food versus fuel debate, although we must be aware that also some non-food agricultural demand like palm oil for cosmetics will be replaced, for which the substitutes may also have GHG emissions.

The share of biofuel land expansion accommodated through consumption **depends on the response of demand compared with the response of supply with respect to price**, i.e. the price elasticity of demand versus the price elasticity of supply. The problem is that neither of them is known very well. It is not only for the feedstock that these elasticities are relevant, but also for other crops, livestock and commercial forestry.

A complication of estimating the **relevant elasticities** is that it is not only about the effect on specific commodities, but also on all crops together and even the effect on livestock production. So, the essential question is to what extent the amount of food, both crop-based and livestock, is reduced as a consequence of biofuel policies. And this share is determined by the difference between the price elasticity of supply and demand.

Most economic models have implicit or explicit **price elasticities** of supply that are equal to or a little bit higher than the price elasticity of demand, implying that between 30% and 50% of ILUC is reduced through consumption changes, although it is a little bit less in the GLOBIOM results.

Although **demand elasticities** from different sources of literature, especially of the US and China, are available on the ERS-USDA website (ERS-USDA n.d.), the empirical foundation is relatively weak, and the database is no longer updated. Recent econometric studies trying to estimate both supply and demand elasticities for agricultural commodities are Roberts & Schlenker (2013) and Berry & Schlenker (2011) using advanced instrumental variable techniques and Haile et al. (2016). They come at supply elasticities around 0.1 and demand elasticities around -0.05, implying that an increase in demand for biofuels of 5% increases the price of the four main staple food commodities with about 35% (Roberts & Schlenker, 2013, p. 2279). They argue that these elasticities are also relevant for the long term, but Baldos and Hertel (2016) argue that long term elasticities are much higher, because there are more adjustment possibilities. Based on Muhammad et al. (2011) they conclude that the price elasticity of demand is somewhere between 0.30 and 0.86, the crop yield elasticity to land rents of about 0.11 (Lubowski et al. 2006), the yield elasticity to price about 0.25 (Keeney & Hertel 2009), and the area elasticity to price about 0.05, having a supply elasticity of about 0.3. This is the same as estimated by Scott (2013). Hertel & Baldos (2016, p. 42) suggest that the area elasticity of supply is about 0.05 after 5 years, and 0.15 after 20 years.

Persson (2016) shows in a literature overview that price elasticities of demand of CGE and PE models are around -0.7, but that they range between 0 and -3.4, while average supply elasticities of the PE models studied are significantly lower at about 0.45, while CGE models have on average a supply elasticity of about 2.48. This is much larger than estimated, but the estimations are normally short term elasticities for specific commodities instead of long term elasticities for crops as a whole.

Long term price elasticities of supply and demand will be different from **short term** elasticities. We may expect that most studies find short to medium term supply and demand elasticities, where for demand it may be expected that the difference with long term elasticities is small, but for supply it may be much larger. In perfect competition price elasticities of supply tend towards infinity, and therefore for example Schmidt et al. (2015) argue that consumption effects should not be taken into account at all. However, for agriculture there is always a restriction on the availability and quality of land, where costs may increase when less suitable land has to be taken into cultivation.

3.3.5. Share accommodated through yields

The **share of production increase accommodated through yields** depends on the **price elasticity of yields compared with the price elasticity of area expansion**. This is the case at all levels, i.e. the biofuel feedstock level, the crop level, and also the agricultural level. However, most models focus on cropland expansion. And in this case the yield elasticity minus the area elasticity is relevant. Gohin (2014) shows that analysis of yield elasticity without consideration of land elasticity is not very meaningful, so the fundamental issue is the relative size of both elasticities.

In interpreting results of price elasticities of yield, we must be aware that they are a combination of differences between yield on new area compared with average area (the marginal yields) that are in some models explicitly included, and changes in yield on current land (that includes increases in double cropping). Searchinger et al. (2008) assume these two effects cancel out, generating a yield elasticity of zero.

In this sense we may define **two types of area and yield elasticities**. The first is the percentage change of area or yield as a consequence of a percentage change in **production**, and the second is a percentage change of area or yield as a consequence of a change **in price**. The price elasticities of area and yield do conceptually not include the effect of marginal yields, but in the practice of estimation the two cannot be disentangled and therefore in practice the ratio between yield and area elasticities both with respect to price and production should be the same. And this ratio is the essence of explaining the share of production increase accommodated through area expansion.

Al-Riffai et al. (2010, p. 92) argue that "there are no robust estimates from the econometric literature because of the complexity of the linkage and the highly fragmented data available for land use in deforested regions, the lack of a continuous time series on local prices, and more importantly, land rent, when they exist" (Malins et al. 2014, p. 91).

The analysis for CARB is originally based on a paper by (Keeney & Hertel (2009) reviewing literature on yield elasticities for the US suggesting a yield elasticity of about 0.25. Berry (2011) and later Searchinger et al. (2015) dispute the size of this elasticity based on a different interpretation of the literature and the fundamental issue of endogeneity of both price, yield and area change implying that the causal relationship from price to area is not proved. Methods that correct for the endogeneity problem like instrumental variables methods should be applied. Furthermore, they criticize that most estimations are on short term effects, where it is plausible that supply responds much less in the short term than in the long term because area expansion or yield increases require investments Berry (2011, p. 7). Roberts and Schlenker (2013) use instrumental variable techniques (using weather as the main instrument) to analyse yield and area elasticities for wheat, corn, rice and soy, where production is defined by total digestible energy content. They find that yield elasticities are small compared with area elasticities. This is developed further by Berry & Schlenker (2011) with estimations not only for the US but also for the whole world, and a more advanced use of instrumental variable techniques. They find a global short-run price elasticity of area of around 0.1, and a price elasticity of yield around zero. Area elasticities are significantly higher, but also far below 1 (around 0.2) for all estimations. Although some yield elasticities are significantly positive, others are significantly negative, showing how unreliable current estimation techniques and data still are (Roberts and Schlenker, 2013).

Miao et al. (2016) estimate the responsiveness of crop yield and area of US corn and soybean to prices and climate by a panel data Instrumental Variable analysis with county fixed effects on US yearly data for 1977-2007. They find a significant own price elasticity of corn yield of 0.23, but not for soybean yield (p. 194), while the price elasticities for area are respectively 0.45 and 0.63, implying that in the short run area expands more than yields, even for corn. The lagged fertilizer price index has a significantly negative effect on corn yield and a significantly positive effect on both corn and soybean area.

Haile et al. (2016) use dynamic panel data estimation techniques on a multi-country multi-crop panel data set and find own price elasticities of yields for wheat, corn, soybeans and rice of 0.166, 0.094, 0.146 and 0.043 and area elasticities of 0.075, 0.069, -.146 and 0.024. They find also significant negative yield effects of increases in crop price volatility, which may be relevant if an inflexible biofuels policy increases volatility in agricultural prices.

The elasticities of the econometric studies are crop specific, or in Berry & Schlenker (2011) specific for the combination of four crops, and because in most studies the estimation approach includes no lagged variables (except as instruments) the results are only short term. The total supply elasticity implicitly in these estimates is so small that it seems to be inconsistent with long term dynamics of agricultural markets where supply elasticities probably tend to very high values.

The IFPRI-MIRAGE study used yield elasticities roughly 10 times as large as area elasticities for most of the world (Searchinger et al. 2015). These are based on area elasticities estimated by Roberts and Schlenker (2013), and yield elasticities based on (Huang & Khanna 2012). But as Searchinger et al. (2015) observe both studies report both elasticities, where in Roberts and Schlenker both elasticities are much smaller than in Huang and Khana, where both have area elasticities that are much higher than yield elasticities. So, this choice seems to be inconsistent, and the implicit elasticities used in the Ecofys study (Valin et al. 2015) seem to be more consistent with the literature from this perspective.

The final recommendation from the CARB Elasticities Values Subgroup (Carb 2011) is to use a yield elasticity of 0.25, taking into account that long term elasticities are larger than short term elasticities because of double cropping and the time lag in introducing new seed varieties or management practices. This argument is still defended as valid, independent of newer econometric studies with lower short term elasticities.

In summary, although some information is available on short term yield and area elasticities for specific crops, it is extremely difficult to get reliable information on long term effects of production increases on yields, because it is almost impossible to disentangle exogenous trends in yields from price-induced yields. Furthermore, the definition of yield increases of crops as a whole is even more difficult to grasp. The empirical evidence on significant yield effects is meagre and mainly short term, implying that the choice of yield and area elasticities is to a large extent based on intuition.

3.3.6. Land use changes because of relocation of production

Yields in different regions differ. Yields of different commodities differ. We have already seen that if maize is substituted for soybean in animal feeding, the required land area may increase. But the issue is broader. For example, Laborde (2011) has a free trade scenario for EU-biofuels where more ethanol is produced from sugar cane in Brazil. This has consequences for land use. It may also be that for example livestock production in Brazil is reduced because of competition of biofuels and that this results in more livestock production in the EU that requires in general less land and has less GHG emissions per kg meat. All these issues are implicit in CGE and PE models, and complicated to trace. In some decompositions in the in depth analysis of some studies we have included an explicit component for this. However, **no study is very explicit on this relocation effect**.

Important aspects of international relocation of production is the **method by which international trade is modelled**. The most common assumptions are the Armington assumption, where current trade flows are the main determinant of future trade, and the minimal cost approach, where the region with the lowest cost (including transportation cost and sometimes some quadratic adjustment cost function) determine the location of additional production.

3.3.7. Location of area expansion

For this section, we base the analysis on Malins et al. (2014) and page numbers refer to this study.

When it has been determined how large the expansion of cropland or total agricultural land is, the basic questions is what **type of ecosystem** is destroyed. The first step is to determine the fraction of cropland expansion into each type of land, the so-called land extension coefficients (LEC) (p. 97). These LEC's may be determined by comparing **satellite data on land cover**, as is done by Winrock-MODIS (p. 97). Their approach is criticized because there is much uncertainty in satellite data. When 5% of area is incorrectly allocated and when this is random, almost 10% of land use changes that are measured may be wrong; this is a multiple of actual real land use changes (p. 97). Miettinen et al. (2012) give more robust results with much more precise satellite data. However, MIRAGE uses Winrock-Modis LECs to allocate land expansion over land categories (p. 98).

A second approach to allocate land expansion is accomplished in models like MIRAGE and GTAP where a CET function determines based on *relative prices* to what extent cropland expands into commercial forest and commercial grassland (p. 97). This approach cannot be used to analyse expansion into pristine land, because for this land no prices are available.

Model results show (p. 98-9, figure 3.11) that they tend to allocate **more land expansion into grassland than into forest**. Forests store more carbon than shrub land and shrub land more than grassland. GTAP used for CARB has only grassland and managed forest to expand in, MIRAGE also has no shrub land, while FASOM uses 25 different forest species types and 18 forest management intensities, so this may influence the carbon consequences.

A third approach to the problem of the allocation of cropland expansion may be the use of a **land allocation model**, where land characteristics per grid cell like rainfall, slope, soil quality, proximity to roads and distance to existing production areas determine the probability of land conversion (p. 98). Malins et al refer in this context to work of the Joint Research Center (Hiederer & Ramos 2010), but also for example the land allocation model CLUE or IMAGE have been used for other studies.

3.3.8. Emission factors

When **land use change is known**, **emission factors** can be applied for the **different land use types**. However, regional and local **variation** may be large, and may also depend on management decisions. Some studies like Schebek (2016) or Overmars et al. (2015) apply explicit land use models for this purpose, but it is not a priori clear that these models are precise enough to allocate crop areas precise enough to generate fundamentally different results. Most studies apply emissions factors to regions, in some cases AEZ's per country. Taheripour & Tyner (2013) provide an overview of a number of land emission factors.

Woods Whole EF's

The most used, but very rough database for estimates of emission factors is developed for IPCC by Woods Hole Research Center (WHCR) and Winrock International (WT), giving data for 10 aggregated regions on vegetation and soil carbon fluxes. They provide information for Forest area and re-growing forest are in million hectares; carbon in vegetation and soil measured in metric tons per hectare; gross carbon uptake by re-growing forests in million metric tons carbon per year; carbon uptake by forest area in metric tons carbon per year. In order to calculate emission factors from these data, assumptions must be made about the percentage of carbon stored in the vegetation which will be released in the atmosphere.

CARB EF's

A more detailed database is developed for CARB, which includes 19 regions and 18 AEZ's. The database includes carbon stocks for a number of pools or sources, including: above and below ground biomass, dead organic matter, soil organic matter, harvested wood products, CH_4 and N_2O emissions, and forgone sequestration. Based on these emissions, factors are derived that are asymmetric: conversion from forest to crop is different from conversion from crop to forest. Taheripour and Tyner observe that the emission factors are based on modelling frameworks with different assumptions making it difficult to check the consistency of the underlying assumptions (p. 7).

TEM EF's

Zhang et al (2009) developed a database of soil and carbon and net primary production on a grid level (0.5′ by 0.5′ spatial resolution). The database is developed by using the TEM model. When these results are aggregated to AEZ-level, they can be compared with the CARB model. While they are quite similar for forests, they differ with respect to pastureland because they use different assumptions on carbon pools and because CARB has more detailed regional assumptions on carbon fluxes.

ANL EF's

This database is to a large extent the same as the WH database, but develops, for the US, a method to calculate changes in soil sequestration depending on crops and management practices. This is especially important for dedicated energy crops that may sequester more carbon than other types of land use (p. 8).

COLE

The Carbon Online Estimator (COLE) developed by USDA (Van Deusen & Heath 2010) assesses above ground forgone carbon sequestration due to biofuel production, depending on management practice, soil decay, soil erosion, and yield improvements.

Taheripour and Tyner compare the different databases and conclude that they can generate quite different results. Reasons include that carbon stocks may differ a lot across locations, estimates of carbon stocks are to a large extent model-based, assumptions of percentages of carbon that is released in the air are uncertain, and it is unclear if all relevant carbon flows and stocks are included.

3.3.9. GHG effect of biofuel policies

In analysing the **effect of a biofuel mandate**, in most models the total effect is roughly the **sum of the effects of all biofuels analysed separately**. So, assumptions about the distribution of different feedstocks over different areas are crucial for the total effects of a biofuel mandate. For example, table 3 explains the difference in GHG emissions between the IFPRI (2011) and the Ecofys study (Valin et al. 2015) for the EU biofuels mandate by differences in the feedstock shares and differences in the GHG emissions/MJ per biofuel type. The rough calculation applied here gives almost the results presented in both studies, and shows that the **differences are almost completely explained by differences in GHG emissions per feedstock, not in the shares applied**.

Table 3 GHG emissions in gCO2/MJ of the extra production in the biofuels mandate split over the different feedstocks. **Source:** Own elaboration based on IFPRI (2011) and Valin et al. 2015.

	Valin share		Laborde share	
	Valin GHG	Laborde GHG	Valin GHG	Laborde GHG
rape+sun	25.9	22.2	19.5	16.7
soy	20.5	7.6	18.0	6.7
palm	49.5	11.6	50.8	11.9
sugar cane	0.4	0.3	3.1	2.5
sugar beet	0.9	0.4	0.9	0.4
cereals	0.8	0.5	2.2	1.3
2nd	-2.3	0.0	0.0	0.0
total	**95.6**	42.6	94.5	**39.5**

3.3.10. Effect of EU environment, climate and agricultural policies on ILUC

In general in ILUC studies **nothing is mentioned about the consequences of other EU-policies on GHG emissions** and therefore it is out of the scope of this study. In this section we mention some possible effects of other policies, some of which are suggested in some of the LUC studies.

Agricultural policies

Some agricultural policies may have consequences for LUC of biofuels. First, farmers get a CAP premium if they keep their land in good agricultural and environmental conditions, even if it is not or marginally used for production. This implies that it will be ploughed preventing carbon sequestration. If this land would be taken into production for biofuels, the loss of carbon stock would be small. Valin et al. (2015, p. xiii) mention that foregone sequestration emissions may not happen because of annual mowing in order to receive CAP money, occasional mowing by smallholders, or extensive grazing.

Second, if agricultural policies promote less intensive schemes with lower yields, then this may reduce the amount of land that can be used for biofuels. Also for example animal welfare regulation, set-aside land policies, and tillage requirements for CAP subsidies may influence the type and amount of land that will be converted.

Third, subsidy policy is an important aspect of agricultural dynamics. Decoupling of subsidies reduced prices and made European feed more competitive with imported feed. Taheripour et al. (2011, p. 11) conclude that in the past crop area in the US was mainly determined by government programs, while currently market forces became more important. The same holds more or less for the EU (Malins et al. 2014).

Fourth, also rural development policy is sometimes focused on improving yields, improving infrastructure which may provide the same type of effects as R&D and extension policies.

Environmental and climate policies

Environmental policies may reduce the opportunities to convert high biodiversity land. In most cases high biodiversity areas are also high carbon areas, so if high biodiversity areas are protected this may reduce the possibilities for land conversion. Legislation and enforcement of legislation in the regions where land use change happens is crucial, and country specific governance is therefore essential.

Environmental legislation in the EU may change options for land conversion. This can potentially force land conversion in areas with low carbon conversion costs, but stricter environmental policies may also drive agricultural production out of the EU with potentially larger GHG effects than it would have had in the EU. GHG emissions may be larger outside the EU because legislation and law enforcement is less or because land management practices are less efficient.

A consistent climate policy that also prices land conversion and GHG sequestration of forests, may reduce ILUC a lot. For example, Valin et al. (2015, p. 39) calculate that a price of 50$ per ton CO_2 would reduce LUC emissions from the EU biofuels policy from 97 g CO_2/MJ to 48 g CO_2/MJ, and if peatland would not be allowed to be converted to 4 g CO_2/MJ. Policies like REDD+ to prevent forest conversion are meant to accomplish some pricing of carbon in forests.

One of the broader issues in the context of biofuels climate policies is to what extent biofuels policy is the most cost-effective method to reduce GHG emissions. Basically, biofuels policies should be taken into consideration in a manner that is consistent with other options for GHG reduction, where the lowest cost options should be chosen.

Trade policies

More flexibility to import biofuels potentially provides an opportunity to reduce GHG emissions. For example, the direct emissions from sugar cane ethanol are much lower than for maize or wheat ethanol, although the indirect emissions depend a lot on land use policy in the producing countries.

More flexibility in trade of crops and livestock in general may change the international relocation of land. If increased biofuels production in the EU is at the cost of other cropland in the EU because other regions are more cost competitive, while these other regions have lower yields or other reasons for larger GHG emissions, the indirect land use effects of EU biofuels may increase with more free trade.

On the other hand, trade policy can also be used as an instrument to force third countries for stricter compliance to environmental regulation. If tariff reduction in free trade agreements are made conditional on environmental policies, then LUC of biofuels may be reduced.

R&D and extension policies

First, research on 2nd generation technologies and technologies to improve yield on marginal land may result in the development of low LUC biofuel pathways. LUC is roughly proportional to direct land use.

Second, research leading to increasing yields for biofuel feedstock will reduce direct land use change and land requirements for non-biofuel purposes, and therefore also indirect land use change.

Third, research leading to a general increase in yields will free land that is needed for non-biofuel purposes and this land is low carbon land without competitive uses that may be used for biofuel production.

What has been said about R&D holds also for extension policies that are meant to spread the knowledge that has been generated by R&D. R&D without diffusion of the knowledge that has been generated is not effective.

3.3.11. Conclusion

Not much is certain with respect to LUC emissions. First, the **dynamics of animal feeding** is very complicated, where in some visions land saving is much more than the percentage of co-products in total production value or weight, but where because of increased palm oil production peatland oxidation may compensate the benefits of co-products. Second, the **reaction of food, feed and other demand for crops** and livestock may be reduced because of price increases, but the size of the elasticities involved, i.e. the price elasticities of supply and demand, are not very well known, and it is especially plausible that long term supply elasticities are much larger in the long term than in the short term, reducing in the long term the price effect of increased biofuel demand as well as the percentage of LUC that may be reduced by consumption reduction. Related with this is the third issue, i.e. the fraction of increased production that is accommodated by yield increases. Recent empirical evidence suggests that the short term price elasticities of yield are small or even zero. However, midterm and long term price **elasticities of yield** may be much larger, because higher prices may stimulate extra investment, including extra R&D, and generate profits that can be used to buy better seeds and more fertilizer and pesticides. However, the same is true for **area elasticities**, where the share of extra production accommodated through extra yields depends on the ratio between the price elasticity of yield and area.

If the amount of cropland expansion is known, it is crucial which **type of land** is converted. We have seen that models differ a lot on what type of land is converted, as for example the IFPRI (2011) study has very small cropland expansions but into high carbon areas, where the Ecofys (2015) study has relatively large cropland expansions in relatively low carbon areas. The last seems to be consistent with the idea that a lot of former cropland is at the moment not used in the EU (where only forest reversion is the carbon cost) and that recently forest land is expanded instead of reduced in the EU.

3.4. Uncertainty

3.4.1. Introduction

Analysis of the uncertainty of LUC factors that are found in different studies is extremely complicated, because it is compiled of a large number of uncertainties in the different models. We have seen that the **basic components** of models like GTAP, MIRAGE and GLOBIOM are the same, but that the **parameters** used are so different that in the end different types of land are converted. Most studies accomplished **Monte Carlo analyses** by varying systematically a number of parameters in the model, and the outcome is in most cases that the spread is very large, while there is no a priori reason why one set of parameters is better than the other, nor that the average values are the most plausible ones (Laborde 2011; Valin et al. 2015; Tyner & Taheripour 2016). Furthermore, it essential to evaluate if you should include **food and feed consumption reductions** as ILUC reductions of biofuels, or not (Searchinger et al. 2015; Schmidt et al. 2015). In this section we discuss a number of representative uncertainty analyses and based on them we evaluate to what extent the range of uncertainty can be narrowed down.

3.4.2. Main Uncertainty Analysis

Ecofys-GLOBIOM analysis

Just as in most important ILUC studies, also Valin et al (2015) do a Monte Carlo sensitivity analysis. They vary demand elasticities, trade elasticities, vegetable oil substitution elasticities, land expansion and yield response elasticities, fraction of oil palm plantations into peatland, the peatland emission factor, co-product protein content, soil carbon impact of straw, yield impact of straw and water availability (Valin, 2015, p. 36). Table 57, reproduced below as table 4, from the report shows the range for which the parameters are varied. For example, demand elasticities are varied between -33% and +50% of their average values, where the variation is for all regions and all products. On the other hand, the trade elasticities are varied between -50% and +100%, where each trade elasticity is varied independently of the other, implying that relative positions of regions and products change. It is interesting that for the land expansion elasticities also is chosen for uncorrelated distributions, implying that the global average land expansion elasticity will not change very much. In contrast, the yield response for feedstock is systematically varied over all regions and only higher yield elasticities are evaluated compared with the standard yield elasticities used. Because yield compared with area elasticity is relevant for the outcome, it is important to compare everything carefully. One has to keep in mind what is varied to be aware of the consequences.

Table 4 Parameter variation used for Monte Carlo analysis. **Source:** Valin et al. 2015.

Parameter	Value range		Distribution assumption	Correlation of parameter between products/regions
	Minimum	Maximum		
Behavioral parameters				
Demand elasticity	- 33%	+50%	loguniform	Correlated across regions and products
Trade elasticity	-50%	+100%	loguniform	Not correlated across regions and products
Water supply elasticity	-50%	+100%	loguniform	Not correlated across regions
Vegetable oil substitution elasticity	-50%	+100%	loguniform	Not correlated across regions
Land expansion elasticity	-50%	+100%	loguniform	Not correlated across regions and land use types
Yield response feedstock	Elasticity model	Elasticity model + 0.2	uniform	Same assumption for all regions
Biophysical parameters				
Co-product protein content	-10%	+10%	uniform	Correlated across regions and products
Soil carbon impact straw	-10%	0%	uniform	Same assumption for all EU regions
Yield impact straw	-4%	0%	uniform	Same assumption for all EU regions Correlated with SOC impact
Peat land emissions factor	27 tCO_2 ha^{-1} yr^{-1}	113 tCO_2 ha^{-1} yr^{-1}	lognormal	Same assumptions for Indonesia and Malaysia
Palm expansion into peat land	12%	54%	lognormal	Same assumptions for Indonesia and Malaysia

The main results of this sensitivity analysis are presented in the main results overview in figure 3 of their report, as shown below. Because the inclusion of foregone sequestration as an ILUC emission source is new and not very certain they separate the GHG emissions without foregone sequestration in blue (with italic numbers indicating the exact value) and the foregone sequestration emissions in green (with the normal numbers indicating total GHG emissions per MJ biofuel). The arrows indicate the range of outcomes between the first and last decile, implying that there are outliers outside these ranges. For some scenarios no arrows are shown because no uncertainty analysis is accomplished.

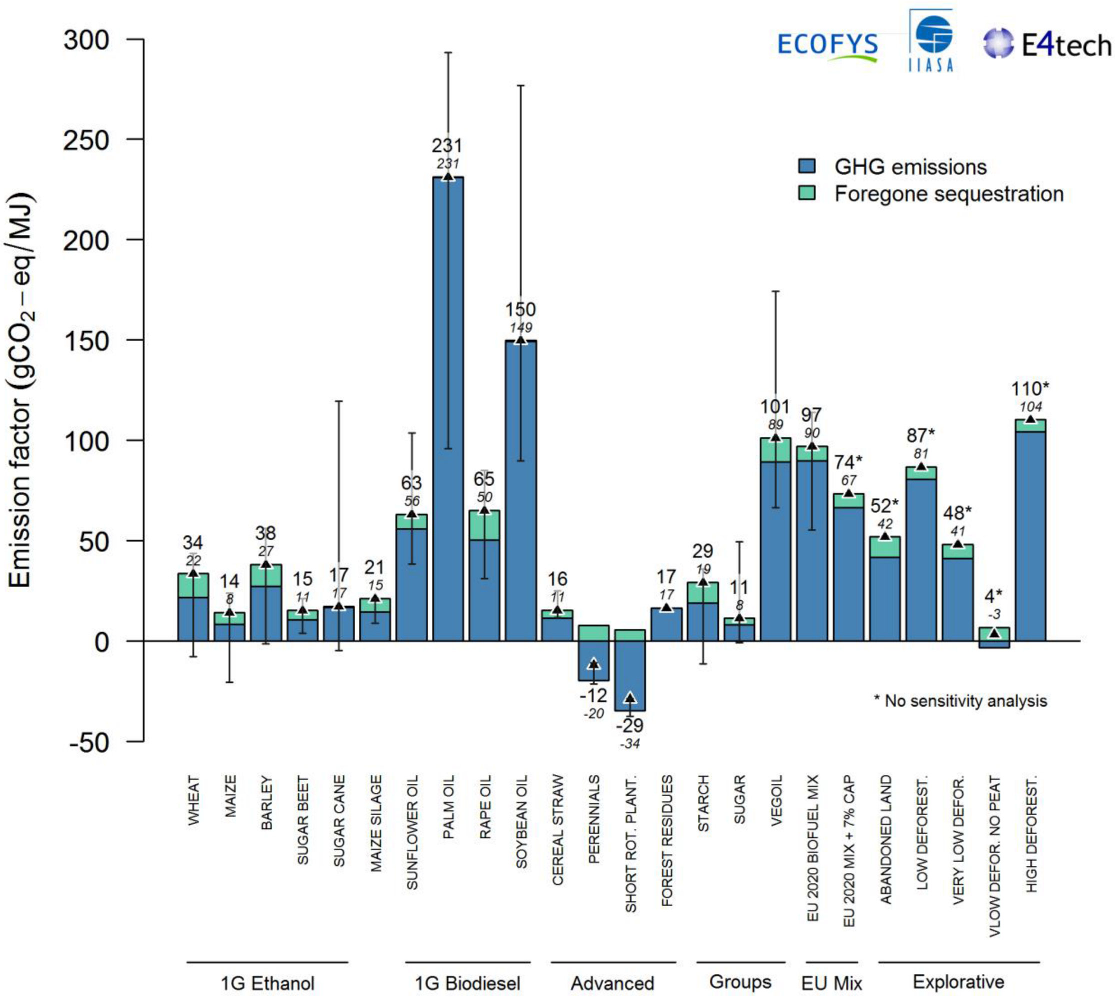

Figure 21 Overview of modelling results: LUC emissions per scenario with and without foregone sequestration and with uncertainty ranges (bars indicate the range within the first and the last decile). **Source:** GLOBIOM.

The results show that the variation in outcomes is large, and that for example for wheat, maize and sugar cane ethanol emissions can become negative. Valin et al remark that while "modelling can be improved with better datasets and better understanding of certain dynamics and interlinkages, uncertainties cannot be avoided." (p. xv)

In annex V of the Ecofys-report, the sensitivity analysis is presented in more detail. A representative idea of what type of information comes out of it, is presented below for maize ethanol.

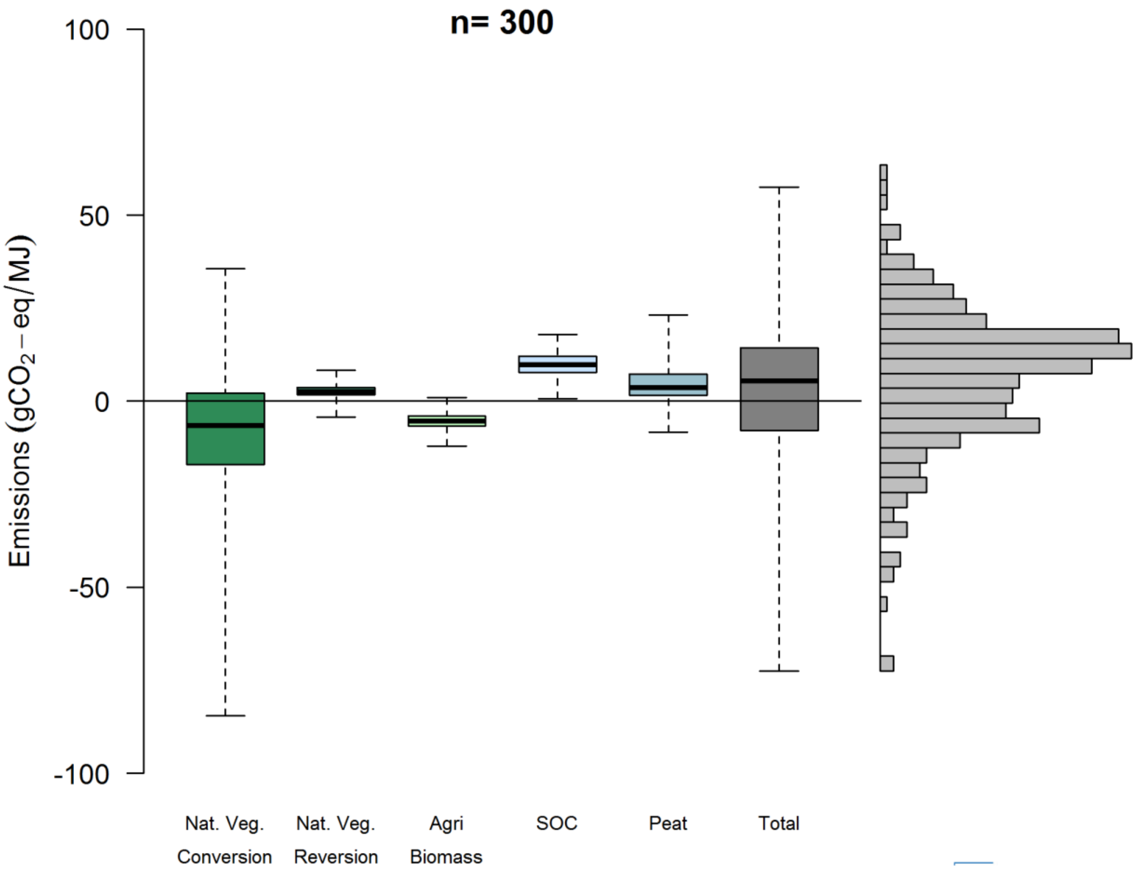

Figure 22 Detail of Sensitivity Analysis. **Source:** GLOBIOM.

The figure shows the average, the 25-75% and the 5-95% uncertainty ranges for different GHG components, i.e. natural vegetation conversion, natural vegetation reversion, agricultural biomass, soil organic carbon and peatland conversion. These variations are the consequence of systematic variation of the parameters, but as it is presented now it is not possible to decompose the uncertainties in the components of LUC as discussed in section 3.3. The result is a total distribution of ILUC factors that is extremely broad, with the 90% interval for maize between about -70 and +60 gCO_2 per MJ, so extremely wide compared with the average of 14 g CO_2 per MJ. We have to be aware that the extremes may also have relevance and that the choice of probability distributions used as well as the intervals of uncertainty investigated are mainly based on intuition because not much empirical information is available.

The conclusion is that important uncertainties remain, because of variability around biophysical values and around causalities assumed by the modelling approach (p. xiii).

Plevin et al (2010, 2015) studies

Plevin et al. (2010) conclude that "lack of data and understanding (epistemic uncertainty) prevents convergence of judgment on a central value for ILUC emissions. The complexity of the global system being modeled suggests that this range is unlikely to narrow down." (p. 8015). There is a lack of consistency in expert judgment on correct parameter values, functional relationships and the efficacy of models to represent the relevant processes. This makes it difficult to model uncertainty probabilistically, and so it may be better to evaluate different scenarios (p. 8016). Despite this, they develop a simple ILUC model that distinguishes the fuel yield, the net displacement factor, the relevant production period where emissions have to be allocated to, emissions factors for forest, grassland and wetland, and the fraction of cropland expansion going into these land cover types.

In their analysis they conclude that the net displacement factor (i.e. the economic part of the analysis) accounts for about 70% of the variance in the emission factors, where it is unlikely that modellers will be able to reduce the uncertainty in this parameter significantly (p. 8019).

In Plevin et al. (2015) a Monte Carlo simulation is accomplished on a combination of the general equilibrium model GTAP-BIO-ADV and the carbon accounting model AEZ-EF. They analyse maize ethanol of the US, sugar cane ethanol from Brazil and soy biodiesel from the US. They characterize parametric uncertainty in the combined model and identify the main parameters that generate the variance of ILUC emissions in the Monte Carlo simulations. Choices on distribution of parameters are based on expert judgment, literature, other model's outputs and sometimes measurement (p. 2659). They find that the economic model uncertainty is the main source of uncertainty, and identify the crop yield elasticity (varied between 0.03 and 0.25, p. 2659) as the main source of uncertainty, contributing between 20% and 50% of the total uncertainty. Marginal yield, emissions from cropland-pasture conversion, the yield elasticity for cropland-pasture, substitution among imports from different regions, and the elasticity of substitution in value-added-energy sub-production are other important contributors to ILUC uncertainty.

With respect to averages emissions are somewhere between 20 and 30 gCO_2e/MJ if consumption is allowed to adjust, and between 30 and 40 gCO_2e/MJ if consumption in developing countries is fixed.

Surprisingly, Plevin et al (2015) vary yields independently of land supply elasticities, where cropland supply elasticities are included as CET substitution parameter between cropland, grassland and commercial forest, with a triangular distribution with width 0.2.

With respect to uncertainty in carbon accounting: available data on biomass are uncertain, estimates of soil carbon fluxes from land-cover change, where the remote sensing used to allocate land cover are highly uncertain (p. 2657).

Uncertainty ranges for corn ethanol, the most studied biofuel feedstock, are roughly in line with other studies (p. 2662, table 3, reproduced below as table 4).

Table 5 Uncertainty Ranges Estimated for ILUC Emission Intensity from Expanded Corn Ethanol Prodution. **Source:** Plevin et al. (2015)

	parameters varied		ILUC emission factor (g CO_2e MJ^{-1})		
model	economic	GHG Acct'g	2.5% value	mean	97.5% value
ILUC-MCS (FNF)	×	×	13	25	42
ILUC-MCS (FF)	×	×	18	33	55
Hertel et al. (2010)[a]	×	×	2	27	52
Plevin (2010)[b]	×	×	21	62	142
ILUC-MCS (AEZ-EF fixed)	×		15	25	41
Laborde and Valin (2011)[c]	×		4[d]	7	8.8[e]
ILUC-MCS (GTAP-BIO-ADV fixed)		×	18	23	29
US EPA (2010)[f]		×	22	30	40

[a]The 95% CI was computed from mean (27) and coefficient of variation (0.46) assuming a normal distribution. [b]Based on the results using uniform distributions. [c]Modified to use a 30-year rather than 20-year amortization period. [d]5% value. [e]95% value. [f]LUC emissions outside the US only, for year 2022. [g]See the SI for a detailed explanation. FF = food fixed; FNF = food not fixed.

In their interpretation Plevin et al conclude that handling of uncertainty depends on the cost of error, and suggest the application of a safety-factor to prevent the wrong decisions (p. 2663).

ICCT's simple model uncertainty analysis

ICCT (Malins et al. 2014) developed a simple macro-ILUC model in order to get a better grasp on the fundamental causes of ILUC. It starts with the situation where all biofuel land would be at the cost of pristine area. The components that reduce ILUC are the reduction in food consumption, the increase in yields, the utilisation of co-products, and the location and type of crops compared with average global yields, and surprisingly included separately the elasticity of area to price, and the carbon stock of new land. Figure 4.2 from the report shows the resulting distribution for corn ethanol using assumptions that are more or less consistent with the Air Resources Board's ILUC estimate and varying for each parameter a best and worst case based, with parameter choices based on judgment and parameter values in the literature (p. 110).

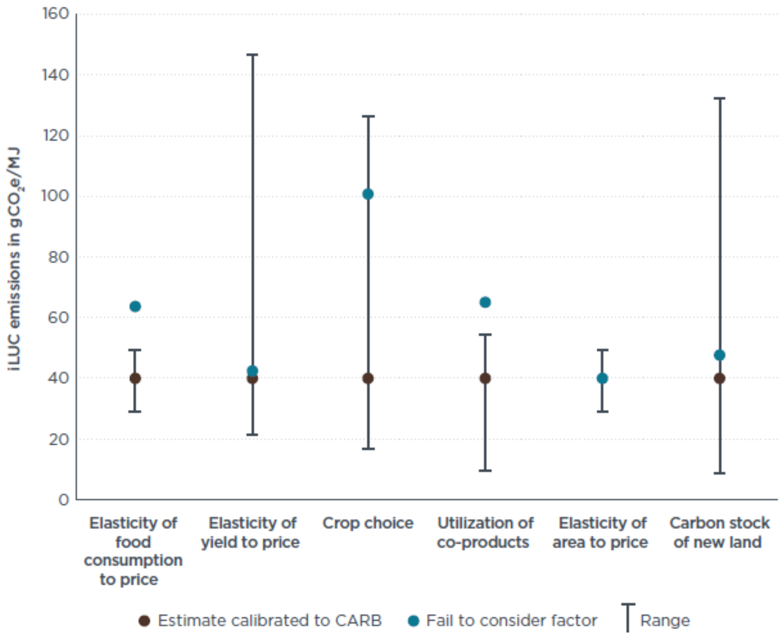

Figure 23 Illustrative model of how assumptions for each parameter affect ILUC results for U.S. corn ethanol. **Source:** Malins et al. 2014.

The analysis shows that varying one of these parameters keeping the others on the default values give very large ranges of ILUC. By varying assumptions on the yield elasticity, the carbon stock of new land or the crop choice ranges can already be very large according to this analysis, with an average of 40 g CO_2 per MJ, where we must be aware that if the consumption reduction would not be taken into account, as Searchinger et al. (2015) suggest, the ILUC factor would already be more than 50% larger.

CARB sensitivity analysis

Tyner & Taheripour (2016) perform a sensitivity analysis for the ILUC analyses performed for the California Air Resource Board (CARB) determining its 2015 ILUC factors for biofuels regulation. For 2015 CARB regulation a new model setting has been introduced by change of economic parameters in GTAP-BIO-ADV and change of emission factor model from Woods Hole Research Center to AEZ-EF model (with emission factors that are sometimes higher and sometimes lower). GTAP changes include a more subtle land substitution nesting structure, where the elasticity of substitution between cropland and pasture can be different from that between cropland or pasture and forestland (p. 16). A new land use category cropland-pasture, i.e. pasture land that used to be cropland in the past in the US or Brazil, is introduced, where it is specifically assumed that conversion from cropland-pasture to annual cropland has half of the emissions of conversion from pasture to cropland, and where it is assumed that conversion to perennial crops has no emissions (p. 3). Also the yield elasticity (YDEL) is reduced. The new version generated a reduction of GHG emissions from 30 to 19.8 gCO_2/MJ for US corn ethanol, from 62 to 29.1 gCO_2/MJ for US soy biodiesel, and from 46 to 11.8 gCO_2/MJ for sugar cane ethanol (p. 2) compared with 2009 regulations.

Tyner & Taheripour (2016) replicate the results generated for CARB and do some sensitivity analyses. Instead of a Monte Carlo analysis Tyner et al perform a number of simulations to indicate the effect of systematically varying some parameters. Tyner et al find that the results can be replicated, and a sensitivity analysis is accomplished on the calibration parameter crop yield price elasticity (YDEL), the yield elasticities and emission factor for cropland-pasture. Also a correction factor for marginal productivity compared with average productivity based on a net primary productivity indicator is investigated, as is the Armington elasticity, i.e. the reaction of trade on international price differences.

With respect to the crop price elasticity of crop yields they show that for small yield elasticity increases the effect on cropland area increase is larger than for larger ones (p. 30), which is logical because with the yield elasticity also the supply elasticity increases, implying that a smaller price increase is sufficient to generate the same effect. As an example, we show their table 3.17 here as table 6 showing the effect of variations of the yield elasticity YDEL in the results of soy biodiesel, implying that a yield elasticity YDEL of 0.05 generates an ILUC factor that is about 50% above the mean value when YDEL is 0.185, while a higher yield elasticity of 0.35 generates an ILUC that is about 25% below the mean.

Table 6 Emission Differences Between the Scenario Averages and Average YDEL: Soy Biodiesel. **Source:** Tyner & Taheripour (2016).

YDEL		$gCO_{2,eq}$/MJ
0.050	Scenario 2	38.25
0.100	Scenario 4	32.51
0.175	Scenario 6	26.70
0.250	Scenario 8	22.78
0.350	Scenario 10	19.09
0.185	Average YDEL	26.09
	Average of 5 Scenarios	27.87

With respect to the marginal divided by the average productivity and the PAEL sensitivity the effect is linear, while increasing Armington elasticities increase ILUC because land use is less in the US and more in the vulnerable areas in the world.

An interesting result is that the size of ILUC depends on the size of the shock. This is caused by the CET elasticities for land. The increase in cropland with an increase in maize ethanol demand from 9.59 to 11.59 billion gallons is 30 % higher than an increase from 1.59 to 3.59 billion gallons, resulting also in about 30% more GHG emissions, i.e. 16.5 instead of 12.5 gCO_2e/MJ.[1]

3.4.3. Possibilities to narrow down uncertainty

Although it is obvious that the tendency between the Searchinger (2008) study and 2012 was in the direction of smaller ILUC factors, there is not such a **tendency** afterwards, because basically **not much new information became available**. Based on criticism the Valin et al. (2015) study tends to have larger ILUC factors than the Laborde (2011) study, very obvious in terms of land use effects per feedstock, and less if calculated in GHG per MJ biofuel. The largest increase in GHG emissions per MJ is for the biodiesel feedstocks. On the other hand, the emissions of especially biodiesel have been reduced by the 2015 revision of CARB. In decomposing the causes of GHG LUC emissions of the Laborde (2011) and Valin et al. (2015) studies (chapter 4) we will see that the mechanisms that explain GHG emissions from LUC can be very different in different studies, even when the ILUC factor is more or less the same.

The fundamental point is that empirical evidence on the components of LUC emissions is very meagre, as we have seen in section 3.3. Supply and demand elasticities are uncertain, and this is even more for area versus yield elasticities, and which land is converted. Also the precise substitution process of biofuel co-products in animal feed is very complex. Perhaps the least uncertain is information on the GHG releases per type of land cover change, but also there the spread is large.

Most studies perform a **type of sensitivity analysis**, mostly in the form of a **Monte Carlo simulation**. However, there is fundamental **uncertainty about the different components**, and therefore it is impossible to define probability distributions objectively.

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[1] Calculation based on table on p. 40.

We therefore must conclude **that ranges of uncertainty have not narrowed down** since 2012, and following Malins et al. (2014, p. 35) that it is **not plausible that they will be narrowed down in the near future**. This is more or less the same conclusion as also found in the Ecofys study: the outcomes are the results of assumed causalities in the model (p. xiii), i.e. the understanding of the agricultural market system (p xv) and on biophysical values. For both it is very difficult to reduce uncertainty.

3.5. Mitigation options and low ILUC biofuels

In this section the scientific evidence related to **ILUC mitigation** and **low ILUC biofuels** is discussed. This is done for the following **four aggregated options** to minimize and mitigate the ILUC effects of biofuels discussed (Wicke et al. 2012; Plevin et al. 2013). See also Deliverable 1.

1. To prioritize the use of **low or zero-ILUC risk feedstock**, such as unused co-products from agriculture and forestry.
2. To prioritize the use of biomass from crops grown on areas that do not compete with food production and that are not used for other purposes, such as **abandoned and unused degraded lands**.
3. To **improve the efficiency** of agriculture, forestry and bioenergy production chains. This includes multi-cropping, integrated food-feed systems and also improved management and R&D in agriculture.
4. To **protect areas** with high carbon stock and/or high biodiversity values, incl. cap on emissions from land use change.

A fifth section is added to discuss the impact of sustainability and certification systems, which often includes strategies to avoid or compensate the ILUC effects of biofuel production.

3.5.1. Low- or zero-ILUC risk feedstocks

An important option for the production of low ILUC biofuels with a relatively large potential is the use of agricultural residues (e.g. straw, stover, manure), forestry residues (e.g. branches, stumps), by-products of the food processing industry (e.g. animal fats) and of the wood processing industry (e.g. bark, sawdust) or other types of waste and residues[2] (e.g. demolition wood, organic fraction of municipal solid waste). Several assessments have been carried out that evaluate the sustainable potential of **residues and co-products** in the EU and other regions, taking into account various theoretical, technical, socio-economic and environmental limitations. These studies indicate that the sustainable potential of residues from agricultural and forestry that is available in the EU at attractive prices can be substantial, although estimates vary considerably (Dees et al. 2017; Elbersen et al. 2015; Spöttle et al. 2013; Pudelko et al. 2013; Khawaja & Janssen 2014; Mantau et al. 2010). At this moment the most important production and use of advanced biofuels in the EU concerns hydrogenated vegetable oils (HVO) produced from used cooking oils, animal fats and other waste oils and fats. Also important is the use of methanol produced from glycerine from first generation biodiesel production. The production of ethanol from wheat straw and saw dust is currently very limited.

No studies were available that evaluate the ILUC effects of the use of used cooking oils, animal fats, other waste oils and fats or glycerine for biofuel production. The use of second generation biofuels produced from lignocellulose residues is, and will likely remain, very limited during the coming years. A few studies have been carried out to evaluate the ILUC effects of the use of residues for biofuels production, which are discussed below.

Overmars et al. (2015) calculated the ILUC effects of ethanol produced from wheat straw in the EU. The ILUC effects are calculated by assigning a part of the ILUC effects of wheat production in the EU to straw, based on the ratio of the value of wheat straw to the value of wheat. The value of wheat straw is 5% of the value of wheat grains. The calculation of the ILUC effects of wheat straw are based on historical data for 2004-2012 about the contribution of yield and area growth to higher wheat production in the EU. The best guess approach of Overmars et al. results in negligible ILUC

[2] From an economic perspective no distinction can be made between waste and residues if the price of both is positive, because both are an output of a certain production process. Therefore, only the term residue is used in this section.

emissions of 2-3 gCO_2/MJ ethanol produced from wheat straw compared to 10-21 gCO_2/MJ wheat ethanol.

A limitation of the study of Overmars et al (2015) is that economic effects of the use of residues are not explicitly considered. Several studies have been carried out that consider economic mechanisms of the use of residues and the resulting ILUC effects.

First of all, the collection, transport and processing of residues requires labour, capital and other inputs. This (additional) demand increases the price of production factors and intermediate inputs, which can result in changes in production, consumption, prices, trade, land use, etc. According to a recent assessment with the GTAP-BIO computable general equilibrium model the use of fertilizers to avoid soil depletion accounts for 23% of the costs of the supply of corn stover, assuming that only 33% of the available corn stover is removed to avoid soil degradation (Taheripour et al. 2013). Results indicate that the production of ethanol from corn stover in the US has only a very small effect on the price of production factors and inputs and also the ILUC effects are negligible. The net ILUC effects are calculated at -1.6 to 0.9 gCO_2/MJ biofuel. The negative emissions are the result of changes in land prices in favour of forestry and marginally causes some reforestation, which seems to be a perverse effect in the model.

Valin et al (2015) assessed the ILUC effects of the use of cereal straw for second generation biofuels production in the EU in case no additional fertilizers are used to avoid a decrease of soil fertility. The analyses are carried out with GLOBIOM, which is a partial equilibrium model of the agriculture and forestry sectors. The use of cereal straw leads to LUC emissions of 16 gCO_2/MJ biofuel. These emissions are the result of a slight reduction in yields of the main commodity (i.e. cereals) in case of overharvesting of straw in regions where high volumes of straw are harvested for other purposes, such as animal feed and bedding. This overharvesting leads to slightly lower cereal yields and to soil carbon and other nutrient depletion. In case the harvesting of wheat straw is limited to a sustainable removal rate of 33-50% then no yield decreasing effects occur and the ILUC effects are consequently zero. The amount of sustainable straw that is available is region depend, so a very careful policy is required.

Valin et al (2015) also assessed the LUC emissions of second generation biofuel production from forestry residues with GLOBIOM. The total net emissions are calculated at 17 gCO_2/MJ biofuel. These emissions are the result of a lower build-up of soil organic carbon.

Potentially more important is the effect that the use of residues increases the profitability of production of the sector that produces these residues. The sale of residues creates revenues for farmers and/or forest owners, which is an incentive to increase the production of the main product. This aspect is evaluated with the MAGNET computable general equilibrium (CGE) model, using the sustainable potential of wheat straw for energy production in the EU in 2030 as a case study (Smeets et al. 2015). The use of wheat straw in the EU decreases the price of wheat in the EU and increases the production and consumption of wheat in this region. The use of land for wheat production in the EU also increases, which is partially compensated by a lower use of pasture land and a reduction of the land area used to produce other grains. Agricultural land use in the rest of the world decreases due to higher exports of wheat and other agricultural commodities from the EU and lower imports to the EU. The shift of agricultural production from the rest of the world to the EU and the high(er) yields per hectare in the EU, results in slightly higher global average yields and in a marginally lower agricultural land use globally. The ILUC effects of wheat straw use in the EU are thus positive, because of the high(er) crop yields in the EU compared to other regions. The net GHG emissions from the use of wheat straw are not calculated, but the global LUC effects per unit biofuel are 5-25% of the impact of first generation biofuels used in the EU as calculated by Laborde et al. (2011), depending on the price of straw. A similar assessment with MAGNET is done based on the large scale use of crop harvest residues for biofuel production in the world in 2030 (Smeets et al. 2016). The land expansion effects of second generation biofuels are estimated at 13% of the impact of first generation biofuels consumed in the EU. This ILUC effect is the result of a higher profitability of crop production due to the additional income from crop harvest residues. Also food consumption increases globally as a result of the use of crop residues.

Based on a similar type of analysis with the GTAP CGE model the ILUC effects of ethanol produced from corn stover in the US are estimated at -11 gCO2/MJ (Dunn et al. 2013). This effect is caused by a shift in agricultural production from other regions to the US, which results in reforestation in Russia. The results of these studies suggest that biofuels produced from residues in regions with high(er) yields can reduce GHG emissions as a result of a shift in crop production from less efficient regions.

Table 7 Overview of estimated LUC and ILUC emissions from the use of residues for biofuel production. **Source:** Own elaboration.

Study	Specification of study	LUC emissions (gCO2/MJ)
Overmars et al. (2015)	Historic approach is used to evaluate the ILUC effects of ethanol produced from wheat straw by assigning a part of the ILUC effects of wheat production in the EU to the straw based on the value of wheat straw to the value of wheat.	-1.6 to 0.9 gCO_2/MJ biofuel
Valin et al. (2015)	Global Biosphere Management Model (GLOBIOM) partial equilibrium model is used to assess the ILUC effects of the use of cereal straw for second generation biofuels in EU in case no additional fertilizers are used to avoid a decrease of soil fertility.	16 gCO_2/MJ biofuel.
Valin et al. (2015)	Assessed the LUC emissions of second generation biofuel production from forestry residues using GLOBIOM	17 gCO_2/MJ
Smeets et al. 2016	MAGNET (Modular Applied GeNeral Equilibrium Tool) is used to assess the impact of revenues from the sale of residues for farmers, which is an incentive to increase the production.	LUC effects of biofuels produced from wheat straw in the EU are negative. Global average LUC effect is 13% of the LUC effect of first generation biofuels consumed in the EU.
Dunn et al., 2013)	The Global Trade Analysis Project (GTAP) model is used to calculate the ILUC effects of ethanol produced from corn stover in the US, which are estimated to be negative from changes in soil carbon.	-11 gCO_2/MJ

The assessments discussed above are all based on recent model based calculations similar to the analyses of ILUC effects of first generation biofuels. In addition to the inherent scientific uncertainties related to ILUC assessment the following aspects are especially **uncertain but potentially important**:

- **Impact on production technology**. The use of residues can have an influence on production technology. For example, a high(er) price of wheat straw can induce a shift from modern to traditional varieties with a lower grain to straw ratio. Modern varieties have a relatively high grain to straw ratio (0.5) comparted to traditional varieties (0.3). The net effect of a shift to traditional varieties would be a decrease in wheat yields and consequently higher ILUC effects.

- **Competition with other applications**. Most assessments discussed above use exogenous assumptions about the sustainable potential of residues as starting point based on other studies. Assessments of the sustainable potential of residues and waste typically first assess potential from the production and processing of crops and wood, using 'multipliers' and 'recoverability factors' that account for various theoretical, technical and ecological limitations. Next, the sustainable potential is estimated by subtracting the use of residues and waste for other purposes than energy production. Estimates of the global potential of residues vary widely as a result of differences in type of residues and waste streams considered and the assumed 'multipliers' and 'recoverability factors', differences in technical and economic limitations and assumptions about the current and future use for other applications. Modelling of the supply and use of residues for different applications through joint profit maximization of crop-residue production and demand and supply interactions can contribute to gain a more detailed understanding of interactions.
- **Soil organic carbon and soil fertility**. To maintain soil quality and to avoid a decrease of the soil organic carbon content and associated GHG emissions part of the residues from agriculture and forestry needs to be left on the field. Several modelling studies have assessed long term effect of straw removal on soil organic carbon (SOC), e.g. Carbon Emission and Sequestration by Agricultural land use (CESAR; Vleeshouwers and Verhagen, 2002) and the CENTURY Soil Organic Matter Model (Melorose et al. 2015; Campbell & Paustian 2015). A study by JRC (Monforti-Ferrario et al. 2015) assessed changes in SOC under different residue collection rates in three scenarios ranging from 0% to 100% removal. The SOC development was simulated by the CENTURY model for the period 2013-2050 for all arable land in the EU. CENTURY calculates carbon input to the soil from plant residues and carbon output from the soil by decomposition of the accumulated organic matter in the soil. Results show that the removal rates for straw at which SOC is maintained vary strongly per location as they are the result of a complex interplay of soil characteristics (especially current SOC levels), climate zones, land cover and the agricultural production itself. The results show that full straw collection leads to a decline in SOC in almost every location with a few exceptions. At the same time it was

confirmed that in areas with higher agricultural yields, larger amounts of straw are produced, thus leading to higher C input into soils which also implies that removal of a part of the straw will not decline the carbon stock in the soil. Overall results show that 50% default removal rates are sometimes enough to maintain SOC, sometimes more straw can be removed and sometimes less. This was also confirmed by the assessment done in the S2BIOM project (Dees et al. 2017) of which the results are presented in the Figure 24 below.

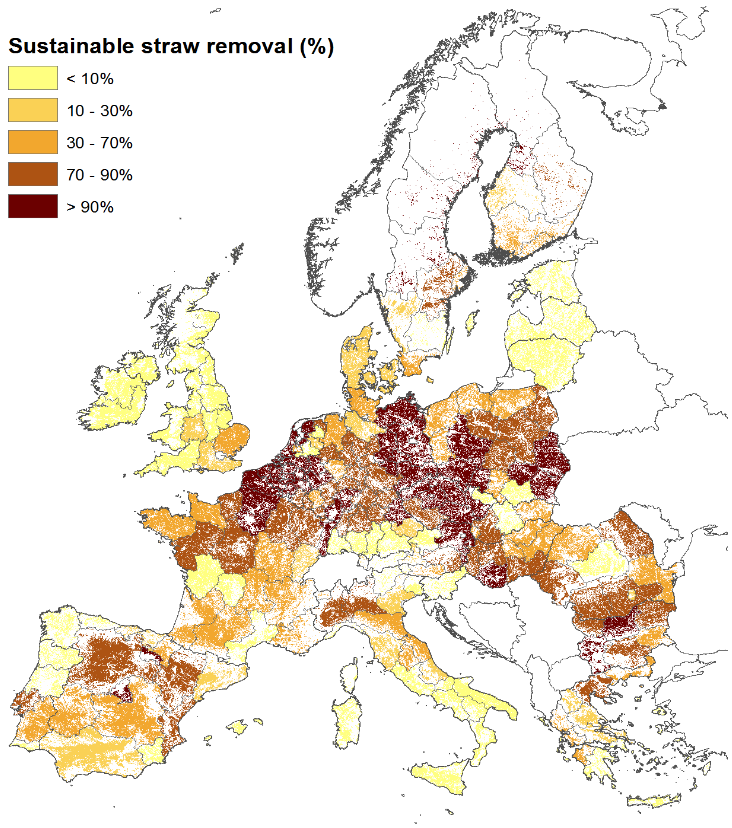


Figure 24 Sustainable straw removal rates in Europe at which no SOC loss. **Source:** S2BIOM project, Dees et al., 2016.

To calculate the sustainable straw removal rates in S2BIOM (Dees et al. 2016) a soil carbon balance at regional (NUTS2 level) was calculated using the MITERRA-Europe model (Lesschen et al. 2012) to provide the input data and the "RothC-26.3" model (Coleman & Jenkinson 1996) to calculate the soil carbon dynamics. Manure and crop residues are the main carbon inputs that were included. SOC decomposition has been included as the only carbon output, other possible C outputs, such as leaching and erosion, are not accounted for. Results show that straw removal rates can be quite substantial in many regions in Europe, even up to 90%, while in other regions they can be very limited, below 10%. Based on the relatively limited number of studies that evaluated the ILUC effects of biofuels made from residues and despite the inherent uncertainties related to these assessments it can be concluded that the ILUC effects of biofuels from residues are most likely very limited. Potentially crucial are changes in soil organic matter, but as long as straw removal rates are based on maintaining the SOC in the soil, the GHG emissions of changes in SOC can be very limited. However, this requires taking account of local sustainable straw removal rates at which SOC is maintained, as these vary strongly per location as they are the result of a complex interplay of soil characteristics (especially current SOC levels), climate zones, land cover and the agricultural production itself.

3.5.2. Feedstock grown on areas that do not compete with food production and that are not used for other purposes

Another option to reduce the ILUC effects of biofuel production is the production of woody or grassy energy crops on **areas that are not suitable for conventional agriculture and do not compete with other land uses**. No information is available about the current use of biofuels in the EU that are produced on areas that are not suitable for conventional agriculture and do not compete with other land use functions. Examples are the use of abandoned farmland, low productive or marginal areas, fallow land and degraded areas. As discussed in Deliverable 1 several studies indicate that the techno-economic potential of biomass production on these areas can be substantial (Chum et al. 2012). However, other assessments suggest that many of these areas are in reality used for various extensive types of agriculture and forestry and also that the biodiversity value is not necessarily low. This means that the real sustainable **potential of crop production on these areas is likely smaller than assumed in these top-down studies**. Furthermore, the low(er) yields of lignocellulose energy crops grown on these areas and lack of infrastructure may limit the **economic attractiveness** of energy crop production.

An important aspect is that the use of areas that do not compete with agricultural land or other uses **avoids ILUC, but still may result in DLUC**. This aspect is analysed in a recent analysis based on the MAGNET CGE and IMAGE integrated assessment model

(Smeets et al., 2016). The use of low productive/marginal areas[3] is assumed not to influence the prices of agricultural commodities and the price of traditional agricultural land is not influenced. In that case the ILUC effects from competition for land are zero, but the DLUC effects are potentially still substantial as plantation expansion leads to conversion of low productive/marginal areas. Based on these assumptions the ILUC effects of global bioenergy scenarios to 2030 are calculated. The scenarios are based on the use of first generation biofuels or second generation biofuels produced from lignocellulosic crops grown on normal quality land that is also suitable for conventional agricultural or on low productive/marginal areas that do not compete with conventional agriculture or other applications. The global land use change emissions are calculated at 12 gCO_2/MJ in case of low productive/marginal areas, compared to 16 gCO_2/MJ for first generation biofuels and 9 gCO_2/MJ in case of plantations that compete with conventional agriculture.

The most important reason for the limited decrease of LUC emissions compared to first and second generation biofuels grown on areas that compete with conventional agriculture is that there is **less competition for land**. Competition for land with agriculture results in higher food prices, lower food consumption and higher agricultural productivity. The lower food consumption and especially the higher average agricultural productivity substantially limits the ILUC effects of biofuels grown on conventional areas. Both effects typically account for more than 75% of the additional supply of biomass for biofuels production in some studies. Both effects do not occur in case low productive / marginal areas are used that do not compete with food production. Results also show an additional benefit of the use of low productive/marginal areas for plantations, which is that the price increase of biomass remains relatively limited to 2030. In case plantations are established that compete with agricultural land then competition for land limits the use of plantations in favour of the use of residues from agriculture and forestry. In this rather comprehensive assessment it should however be mentioned that it does not take into account the potential SOC accumulation because of the production of energy crops on marginal

[3] In IMAGE, these marginal lands are non-forest, non-agricultural lands, with a natural vegetation cover of grass or shrubs

lands. Be also aware that the marginal lands could also have been used for the production of other commodities like paper pulp that may reduce production of these commodities elsewhere reducing the pressure on pristine areas or releasing agricultural land.

Valin et al. (2015) assessed that the ILUC effects of biofuels produced from grassy energy crops (miscanthus and switchgrass) and from short rotation coppice in the EU using the GLOBIOM model. In the case of biodiesel production **from switchgrass and miscanthus** about one third of the additional land area is projected to come from abandoned agricultural land, the remaining from natural vegetation. The net ILUC effects are calculated at -12 gCO_2/MJ biodiesel. These negative emissions are the result of the sum of foregone sequestration on abandoned cropland, conversion of natural vegetation to agricultural land and an increase of soil organic carbon and biomass content of crops. So, it is basically assumed that the carbon value of the biofuel feedstock is higher than the carbon value or carbon sequestration potential in the original vegetation. To underpin this type of analysis, several studies highlight the potential of woody and grassy energy crops to restore contaminated soils, although the potential is determined by local and regional conditions (Nsanganwimana et al. 2014).

In the same study also the ILUC effects of advanced biodiesel produced from **short coppice plantations** are evaluated. The total net ILUC induced GHG emissions are -29 gCO_2/MJ biodiesel. Almost all ILUC effects are projected to occur in the EU. About 18% of the increase in plantation area comes from a reduction of the abandoned cropland area. The negative ILUC effect is caused mainly by carbon sequestration in short rotation coppice plantations, which are only partially undone by foregone carbon sequestration on abandoned cropland.

The use of **abandoned agricultural land** is also explored by Valin *et al.* (2015). In that scenario it is assumed that specific policy incentives are introduced to stimulate the use of abandoned land in the EU. Without such stimuli a large part of the biomass will be imported from abroad. The abandoned agricultural land scenario is implemented in GLOBIOM by a combination of abandoned land restoration incentives combined with a ban on biofuels made from palm oil, soybean oil and sugar cane. This scenario reduces the ILUC emissions of biofuel use in the EU from 97 gCO_2/MJ to 52 gCO_2/MJ biofuel. This reduction comes partially from a higher use of abandoned land and partially from lower use of biofuels from palm oil. The disadvantage of this experiment is that it combines a lot of changes in biofuel production; it doesn't separate the effect of the use of more abandoned land.

Based on the above it can be concluded that **using various types of unused land is a potentially promising option to avoid ILUC effects of biofuel use**. The **net ILUC effects are however uncertain**. Negative emissions occur in case of restoration of degraded areas, although foregone carbon sequestration and carbon losses from DLUC of natural vegetation can still be significant.

3.5.3. Increasing the efficiency of agriculture, forestry and bioenergy production chains

Several studies emphasize the importance of **improving the efficiency of agriculture** to avoid the conversion of natural vegetation and associated undesirable effects on biodiversity and emissions from ILUC. However, as discussed elsewhere in this report the endogenous yield increase from higher demand for crops for biofuels production is **insufficiently large to avoid an expansion of agricultural land and ILUC effects**. Policies aimed at increasing the productivity of crop and livestock production can have a large effect on land use and on the ILUC emissions of food and biofuel production. However, similar to the protection of natural vegetation these effects cannot be allocated to the production of biofuels only, unless these policies are implemented to offset ILUC effects of biofuel use.

Estimates with the MAGNET CGE model suggest that compensating ILUC effects through higher investments in R&D in agriculture can be realised at **limited additional costs,** both in the EU but especially in developing regions (Kristkova et al. 2016). They estimate the costs of R&D investments to avoid negative LUC effects at 0.4 to 0.6 $/GJ biomass which low compared with a price of about $10 per GJ biomass. However, the return on these investments in R&D depends on a limited body of empirical knowledge, where the investment could also be done without the biofuels.

Furthermore, as far as higher agricultural yields also generate a general cost reduction, they may also lead to **higher food consumption**, depending on the price elasticities of demand and supply. Food price and food consumption effects are less costly to compensate, i.e. investments in R&D to avoid ILUC effects of biofuels reduce food prices and increase food consumption.

Furthermore, many biofuel **certification systems** include criteria for good agricultural practice and to optimize crop production, but few explicitly quantify the low ILUC potential. In 2016 Ecofys published a methodology to identify the supply of additional biomass realised by higher crop yields through improved inputs and management practices, including better fertilisation, irrigation, seeds and equipment (Peters et al. 2016; see also section 3.5.5**Error! Reference source not found.**).

ILUC effects can also be reduced by **stimulating higher productivity in other sectors competing for land**. This was for example shown in the study by Lapola et al. (2009) who showed that if livestock density would increase with 0.13 head per hectare instead of the baseline 0.09 head per hectare this could completely avoid the indirect land-use changes caused by biofuels production in Brazil. Lapola et al therefore suggest that a closer collaboration or strengthened institutional link between the biofuel and cattle-ranching sectors in the coming years is crucial for effective carbon savings from biofuels in Brazil.

Integration of food and feed systems is another possible strategy to increase the productivity per hectare land. However, existing integrated systems are mainly based on the use of co-products from biofuel production (e.g. the use of residues from sugarcane ethanol production as animal feed).

As a comment of all approaches to invest in R&D or intensification for higher yields is that these policies are **as useful for food production as they are for biofuel production**. If investments are profitable to compensate for biofuels production they are also for other production purposes. It is basically a conditional sale of biofuels and R&D or investment in higher yields.

3.5.4. Protecting areas with high carbon stock and/or high biodiversity values

An important aspect is that policies to avoid conversion of natural vegetation are not necessarily the result of the use of biofuels or policies that stimulate the use of biofuels. In other words, the **benefits of protection of natural vegetation and lower ILUC emissions from food and biofuels production cannot be allocated to the production of biofuels only**, unless these policies are implemented as part of the policies that stimulate the sustainable production and use of biofuels. Moreover, the **protection of natural vegetation may limit the ILUC emissions of biofuels**, but this may also lead to a **trade-off with higher food prices and higher impact on food consumption**.

Winchester & Reilly (2015) conclude that global bioenergy use results in deforestation if no costs are associated with emissions from land use change. As regions are linked via international agricultural markets, irrespective of the location of bioenergy production, natural forest decreases are largest in regions with the lowest barriers to deforestation.

Valin et al. (2015) evaluate the impact of global policies to limit deforestation and peatland drainage on ILUC emissions of biofuels. A price of USD 50/t CO_2 emissions from deforestation reduces the LUC emissions of the EU 2020 biofuel mix from 97

gCO_2/MJ biofuel to 48 gCO_2/MJ. If also emissions from peatland are avoided then the overall LUC emissions from biofuels used in the EU would further decrease to 4 gCO_2/MJ. The share of crops used for biofuels production that is diverted from the food and feed sector is however higher in these low deforestation scenario compared to the default scenario of biofuel use in the EU. We must also be aware that the results of such an exercise depends on the way such a carbon price is implemented and assumptions on cost and benefits of deforestation for the people who are involved in the deforestation or the use of peatland.

In summary, measures to reduce deforestation and peatland drainage are only effective if they are **accommodated for all land using sectors**, and not only the biofuel sectors. The protection of high carbon areas reduces ILUC, but at the same time increases agricultural prices.

3.5.5. Sustainability and certification systems

Most existing sustainability and certification schemes **do not explicitly deal with ILUC**, but several include **measures that avoid undesirable DLUC effects** and **indirectly avoid or compensate ILUC effects**.

Recently, Ecofys published a study 'Methodologies identification and certification of Low ILUC risk biofuels' (Peters et al. 2016) in which two methodologies are presented to identify the supply of additional biomass by means of 1) **higher crop yields** through improved inputs and management practices, including better fertilisation, sowing practices, crop rotation, crop protection, pollination, harvest, and precision farming or (2) **expanding agriculture on previously non-agricultural land with low carbon stocks and low biodiversity value**. Both issues were already discussed above, but we will investigate the proposed methodology a little bit more in-depth.

The key starting point of the approach is that of **'additionality'**. The economic operator (which can be one farmer or a group of farmers or a whole region) needs to prove that additional biomass production that can be certified as 'low ILUC' comes above what is produced in a baseline situation and that the incentive to increase yields or take unused lands in production comes from for an additional non-food biomass demand.

The identification of biomass supply from **higher crop yields** is based on a comparison of the development of the actual productivity compared to the trend line development of crop yields. If feedstock producers can demonstrate that yield increases are above the trend line then the additional production is qualified as low ILUC feedstock. Crucial thereby is that the higher productivity does not occur in absence of biofuel production and can be attributed to improved management.

A potentially problematic aspect is that the **variability and** changes in productivity are largely determined by weather and climate, so this can only be proven with long term historic yield data (10 years) which are not always available in all regions and at a reliable spatial resolution. At a farm level it is important to ensure that reliable long term yield data are provided by the Economic Operator.

The study supports certification for strategies that focus on increasing yields in existing crops, but also through adopting **cropping rotations**, and through **multi-cropping practices** such as double cropping and intercropping. Because of this a more complex approach is presented to analyse the yield increase above the trend line. This approach cannot focus on the main product yield, but should take account of the protein content of the total harvested yield or the specific useful components of the biomass such as starch, sugar and fats or, in case of whole crop use (e.g. forage crop) in forage units per tonne fresh matter. So, the fundamental idea of this approach is that a farmer who likes to produce biofuels is required to invest also in yield increases.

There are however a couple of **additional considerations** needed that ensure how solid the **increased yield approach** is:

1. It is extremely difficult to prove that a farmer would not have **implemented the measures** to increase yield if biofuels would not have been produced. It would also be valuable to investigate how farmers alter the management of their total farm and whether the focus on yield increase in certain fields and crops will not come at the expense of yield reduction in other food and feed crops.

2. There is one concern about a management practice that has specifically not been listed as a measure to increase yield and that is the introduction or **improvement of irrigation practices**. The study does not allow for this strategy to be part of the ILUC mitigation practices, but what if it accompanies the practices that are listed? If this happens, it can lead to an increase in unsustainable irrigation practices depleting sweet water resources in arid regions. It will also be very challenging to determine which part of the yield increase is related to the listed management adaptations and which from the additional irrigation measures. After all, yield increases can only be reached if limitations are solved for the diversity of inputs required. More nitrogen use can only result in a yield increase if there is also enough water.

3. Several strategies for crop yield increase are proposed in the Ecofys study, but the one for introduction of **agro-forestry systems** is not included. Since agro-forestry systems have large sustainability advantages, particularly in more arid regions of the world and in relation to carbon accumulation below and above ground (Jose & Bardhan 2012; Nair et al. 2009), it is recommended to also include these and make an inventory of GHG mitigation effects.
4. It should be acknowledged that the **calculation of the trend line**, will be complicated and data intensive and remains very challenging for an economic operator. It is questionable whether economic operators in all regions of the world will have similar access to data and enough knowledge to apply for the 'low ILUC' certificate based on yield increase.

The second strategy of **growing biomass crops on unused lands** requires economic operators to prove the absence of other provisioning services on this land in the last 5 years in order to ensure 'additionality'. Furthermore, it is required to comply with EU RED sustainability criteria for biofuel production when the land is taken into production. Much attention in the report is then paid to techniques and data for verifying the unused land status. The approach presented is interesting, but still has several **open issues**:

1. High expectations are placed on the technique of Normalised Difference Vegetation Index (NDVI) and Enhanced Vegetation Index (EVI) enabling the observation, using **high spatial and temporal resolution remote sensing data**, of vegetation development and management disturbances. These techniques are indeed developing rapidly with increased earth observation data availability, but still have many limitations and need improvements, particularly in relation to validation of the observed patterns with real measured/statistical land use management data.
2. One can doubt whether the **5 year unused land status** is long enough to rule out certain very extensive grazing practices which involve the grazing of land at intervals of more than 5 years. The function of grazing on these lands, even though at long intervals, can be very important for maintaining specific biodiversity values which are also difficult to measure alone with NDVI approaches. For evaluating the biodiversity values of these lands species and habitat mapping data are crucial, particularly where it involves lands with permanent grassland cover. Such cover can only be maintained in some stable grazing situation which prevents shrub invasion and thus indicates towards

grazing activities of livestock or wildlife at regular although very long intervals. Communal lands can also remain unused at very large intervals especially in arid areas, but this does not mean they are not used at all by livestock or wild animals (Elbersen et al. 2014).

3. In the Ecofys study unused lands can only be converted to crop land for biomass if not falling in the category of **high carbon stock land**. This aspect is not further discussed in the report, except that it is mentioned that this aspect can be evaluated by a certification body through soil samples. This approach seems to be rather limited as high carbon stock areas need further definition to prevent net loss of carbon through producing biomass on unused lands. In the RED high carbon stock land is referring to wetlands and continuously forested areas. It should however be realised that unused land, if left undisturbed for a long time, can build up a lot of above and below ground biomass. When converting this land into a plantation with annual or perennial crops there is a certain time period needed to compensate for this loss of carbon. This needs further evaluation in order to come to precise requirements for the mitigation capacity of the new cropland. It is likely that compensation will much sooner be reached with a perennial than with an annual crop (Lewandowski et al. 2015; Lord 2015; Masters et al. 2016; Monti & Zegada-Lizarazu 2016; Mbonimpa et al. 2016).

Last but not least, there is the issue of **length of the 'low ILUC' certification period** which still leaves uncertainty about several sustainability risks. The Ecofys study proposes that once the certification is acquired it remains certified for 10 years both for the yield increase and the unused land strategy. At the same time the study acknowledges that taking a 10 year period is difficult to defend and has moral and sustainability aspects as in these 10 years global food demand increases too. Within this 10 years' time the additional yield above trend line and biomass production on new land could change the status from 'ILUC free' to 'competing with food production' given increased food demands. The Ecofys study therefore indicates the need to set a cap on the amount of biomass that can be certified. How this cap is to be determined requires further evaluation studies that have not been published so far and are not addressed in the Ecofys study, but do need further consideration. Other sustainability aspects that have not been mentioned in the Ecofys study are the following:

1. Is **10 years** not too short in relation to the build-up of soil carbon and total above ground biomass in the case that unused lands are converted to perennial plantations? **Perennial plantations** with e.g. miscanthus or different types of

short rotation coppice usually have a lifetime of more than 10 years. If a plantation is finished before it has reached its maximum carbon mitigation potential it is a missed opportunity. There is a need to further investigate the relationship between plantation lifetime of different perennials and mitigation potential. There is already much work published in relation to this aspect (e.g.Monti and Zegada-Lizarazu 2016; Lewandowski et al. 2015; Larsen et al. 2014; Fagnano et al. 2015).

2. A lifetime of **10 years** for the low ILUC certification may also be too short for the economic operator to earn back **costs and make a profit**. This may well bring down the take up of the measure. Further evaluation studies are required to cover this aspect and identify land use practices that are most economic and carbon efficient within a certain lifetime. Based on the outcomes the low-ILUC certification can enhance the most sustainable and economically valuable options for bringing unused lands into production.

Overall the low ILUC certification approach presented by Ecofys seems be circumvented with many unsolved problems, and may also be doubted from a logical point of view because it is not clear why land that can be used fruitfully for biofuels could not be used for other purposes like producing paper pulp or even agricultural production.

Even if the approach would make sense, the implementation of the yield strategy needs further refinement particularly regarding the practical calculation of trend line yields and reliable data availability in all regions of the world and risk for unsustainable increases in irrigation water consumption. The evaluation of the unused land status and the duration of certification of 10 years still have many open ends which have to be evaluated further. One should consider at the same time that limiting agricultural expansion to previously non-agricultural land with low carbon stock and low biodiversity value are most effective in avoiding ILUC effects if extended to all land uses including food and feed.

In general it can be concluded that the **certification of low ILUC and ILUC free biofuels is unlikely to be able to avoid all indirect effects**. **Additional measures**, beyond the scope of certification, are therefore needed, such **as integrated land use planning including territorial policies.**

3.6. A broader perspective

3.6.1. Biofuels, food, and fossil fuels

In order to put the standard approach to ILUC analysis we developed figure 25 to illustrate the **relationship between policy and effects of these policies** in an attempt to answer the questions which GHG emissions should be allocated to the biofuels, and which are the consequence of the biofuels policy but should not be allocated to the biofuels.

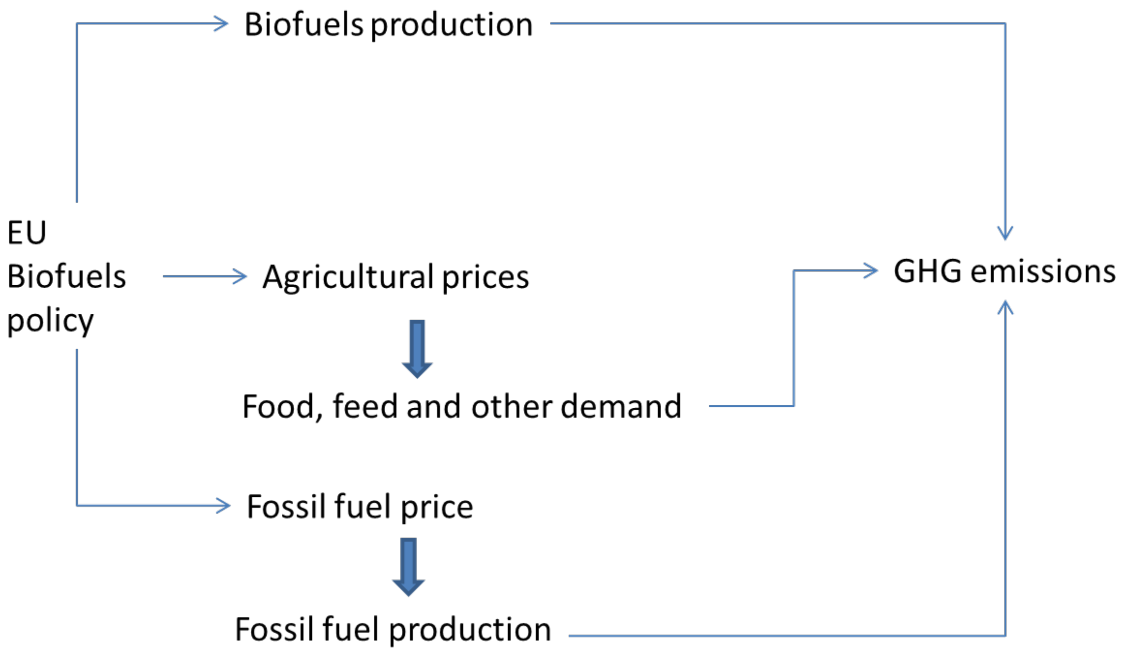


Figure 25 EU Biofuels Policy impact on GHG emissions. **Source:** own design

Biofuels policy has as a target to substitute fossil fuels into biofuels. This accomplished by forcing blending of biofuels into transport fuels, and this increases **biofuels production**. The production of biofuels generates extra demand for agricultural commodities and this generates another consequence of biofuels policy, i.e. an increase in **agricultural prices** reducing consumption, animal feed and other agricultural demand. This **reduced food, feed and other demand** reduces GHG emissions, but this is caused by the reduced demand for feed and food, not by biofuels production. In the same way, blending of biofuels in the EU reduces global demand for **fossil fuels** and therefore reduces fossil prices stimulating fossil fuel demand. This increased fossil fuel demand outside the EU increases GHG emissions. The cause of these emissions is the increased fossil fuel production. So, biofuels policy increases biofuels production as an intended consequence, but reduces fossil fuel demand less than the energy value of biofuels production, and this implies GHG emissions. In the same way the policy generates reduced food consumption and with this reduces GHG emissions. **All are the consequence of the biofuels policy, but the GHG emissions must not be attributed to biofuels production, but to the other changes in demand generated by the biofuels policy.**

This vision is **contrary to the standard way that ILUC factors are calculated**, where reduction of GHG as a consequence of reduction of food demand is seen as a reduction in ILUC of biofuels. Searchinger et al (2015) suggest that this should at least be made explicit, but you could take a stronger point of view in the sense that in the evaluation of the GHG emissions of biofuels the reduction of GHG as a consequence of a reduction in food reduction should not be taken into account, just as also the rebound of fossil fuels is not taken into account. Both are a consequence of biofuels policy, but are not the consequence of biofuels production itself. Potentially, the effect could be corrected by taxes that compensate for price changes because of biofuel production, although in practice this will not happen.

Although not included in figure 25 we could take the analysis a step further in that the yield effect of higher prices of agricultural commodities should also not be considered as an ILUC benefit. The yield increases are the consequence of investment in R&D, changes in fertilizer, pesticide and water use, and different agricultural management systems. So, the benefits should be allocated to them.

An alternative approach could be that you evaluate all the effects of biofuel policies and its influence on GHG emissions. This implies that you not only include the reduction in consumption, but also the increase in fossil fuel production as a consequence of the rebound effect and for example changes in emissions from fertilizer use and changes in the wood products industry and the industries to which these industries deliver. So, there is no reason to separate out the ILUC effect from other effects.

In contrast with this, land savings because of biofuel co-products should be attributed to the biofuels, because these are directly caused by the biofuel production. The increase or reduction in meat production that may be caused by lower or higher feed prices as a consequence of biofuels, again, should not be included as a cost or benefit of biofuel production, because the final cause is a change of meat consumption.

In summary, from a conceptual point of view it is not easy to find out **what type of ILUC reducing factors should be taken into consideration**.

3.6.2. N_2O emissions

Intensification causes increases in GHG emissions, and therefore these emissions should be taken into account when analysing ILUC effects of biofuels. Plevin et al. (2015b, p. 2657) discuss the US EPA analysis where the effect of on-farm energy use, fertilizer use and N_2O emissions and CH_4 emissions from livestock and rice production is investigated. In tables S7 and S8, reproduced below, they show the effect of the inclusion of **N_2O and CH_4 on ILUC emission factors** in some scenarios, implying an **increase in GHG emissions** of about 10 gCO_2eq/MJ, although the results seem not be completely consistent.

Table 8 Summary of results for ILUC emissions (g CO2e MJ^{-1}), not including changes in emissions of methane (CH_4) or nitrous oxide (N_2O). FNF=food consumption not fixed anywhere; FF=food consumption is fixed in non-Annex I countries **Source:** Plevin et al. (2015b, p. 2657)

Experiment	Mean	2.5 percentile	97.5 percentile
Corn ethanol, FNF	25	13	42
Corn ethanol, FF	33	18	55
Cane ethanol, FNF	25	10	43
Cane ethanol, FF	36	17	59
Soybean biodiesel, FNF	25	13	43
Soybean biodiesel, FF	38	21	62

Table 9 As described in previous table, except including changes in emissions of methane (CH_4) and nitrous oxide (N_2O) **Source:** Plevin et al. (2015b, p. 2657)

Experiment	Mean	2.5 percentile	97.5 percentile
Corn ethanol, FNF	36	26	52
Corn ethanol, FF	46	33	68
Cane ethanol, FNF	18	4	35
Cane ethanol, FF	33	15	56
Soybean biodiesel, FNF	32	21	49
Soybean biodiesel, FF	48	32	71

3.6.3. Impact assessment of biofuels policy

It is obvious that **biofuels policy has more consequences than only ILUC**. In section 3.6.1. it has been suggested to exclude consumption effects from ILUC, while if you include consumption effects it seems logical to include also other price effects like the rebound effect as a consequence of price reduction of fossil fuels. In evaluating the total policy, a lot of other issues are relevant, some of them may be worth mentioning.

First, it is obvious that to calculate the total effect of the biofuels policy both **direct and indirect emissions** should be taken into account, where if intensification is included also N_2O and CH_4 emissions should be considered. Second, the **consumption effect** is relevant, with both positive consequences as less waste and less overconsumption of food, and negative consequences like increased hunger and nutrient availability. Third**, environmental effects**, like extra pollution of ethanol compared with fossil gasoline when used in the car and the losses in biodiversity.

Fourth, as far as biofuels are meant to reduce GHG emissions the **social cost** of reducing GHG emissions through the production of biofuels should be compared with other options to reduce GHG emissions. If all least cost options are chosen in a consistent climate policy, then a price of GHG emissions would automatically generate information on which biofuels are profitable from an economic point of view.

Fifth, the targets of biofuels policy may **not only be greenhouse gas emissions**, but also for example a measure to help the agricultural sector or to reduce risks of fuel supply. As far as this is the case, the policy should also be evaluated from these perspectives.

In summary, the analysis on ILUC in this report is only a subset of the issues that are relevant for the evaluation of biofuel policies. The suggestions above are some other issues that have **to be taken into consideration**. However, they are not part of the ILUC reporting requirements and are therefore not further discussed in this report.

3.7. Conclusion

Not much is certain with respect to LUC emissions. First, the **dynamics of animal feeding** is very complicated, where in some visions land saving is much more than the percentage of co-products in total production value or weight, but where for because of increased palm oil production peatland oxidation may compensate the benefits of co-products. Second, the **reaction of food, feed and other demand for crops and livestock** may be reduced because of price increases, but the size of the elasticities involved, i.e. the price elasticities of supply and demand are not very well known, and it is especially plausible that long term supply elasticities are much larger in the long term than in the short term, reducing in the long term the price effect of increased biofuel demand as well as the percentage of LUC that may be reduced by it. Related with this it the third issue, i.e. the fraction of increased production that is accommodated by yield increases. Recent empirical evidence suggests that the short term price elasticities of yield are small or even zero, but midterm and long term price elasticities of yield may be much larger, because if higher prices stimulate extra investment, including extra R&D, and generate profits that can be used to buy better seeds and more fertilizer and pesticides price elasticity of yield may be larger in the long term than in the short term. However, the same is true for area elasticities, where the share of extra production accommodated through extra yields depends on the ration between the price elasticity of yield and area.

If the amount of cropland expansion is known, it is crucial which type of land is converted. We have seen that models differ a lot on what type of land is converted, as for example the IFPRI (2011) study has very small cropland expansions but into high carbon areas, where the Ecofys (2015) study has relatively large cropland expansions in relatively low carbon areas.

As a consequence of all the **uncertainties** in the components of LUC emissions it is very **difficult to narrow them down**. The analysis by ICCT (Malins et al, 2014) shows that even if you analyse the variation for each component then plausible ranges for LUC emissions can be very broad. If you combine all types of uncertainty, as is done in most sensitivity analyses, for a lot of biofuels the spread is also very broad, where it is however difficult to trace down the causes of the probability distribution of the emissions. For this reason, doing a sensitivity analysis by just varying one parameter at a time as Tyner and Taheripour (2016) do, may be more informative.

Normally studies reach levels of emissions for standard biofuel policies that imply relatively minor GHG savings, and increases in GHG emissions if reduction of non-biofuel consumption is not allocated to the biofuels. So, the question is to what extent **mitigation options** to reduce LUC emissions are available.

The first strategy focuses on **low ILUC feedstocks**. One type of low ILUC biofuels is to produce them from co-products like straw and forestry residues. There seem to be opportunities from an ILUC perspective, but one has to take into consideration that:

- Harvesting residues may be at the cost of organic soil carbon
- Harvesting residues may provide incentives to switch to techniques with lower productivity for the main products
- Harvesting residues for biofuels may be at the cost of using them for other purposes.
- Harvesting residues for biofuels must also be cost effective

The second strategy is to grow feedstocks **on marginal lands**, i.e. land that is not used for other purposes. When perennials are used on degraded or low carbon land that would not be used otherwise, the carbon value of the biofuel feedstock may be higher than the carbon value or carbon sequestration potential in the original vegetation, generating negative emissions from land use change. However, be also aware that the marginal lands could also have been used for the production of other commodities like paper pulp that may reduce production of these commodities elsewhere reducing the pressure on pristine areas or releasing agricultural land.

A third strategy is to **increase yields**. Several studies suggest that investment in R&D and extension services has high returns. However, if you require these investments to certify biofuel production, it is basically a conditional sale. So, if these policies are useful for biofuel production, why wouldn't you apply them also to food production?

A fourth strategy is the protection of **areas with high carbon stocks**. An important aspect is that policies to avoid conversion of natural vegetation are not necessarily the result of the use of biofuels or policies that stimulate the use of biofuels. In other words, the benefits of protection of natural vegetation and lower ILUC emissions from food and biofuels production cannot be allocated to the production of biofuels only, unless these policies are implemented as part of the policies that stimulate the sustainable production and use of biofuels. Moreover, the protection of natural vegetation may limit the ILUC emissions of biofuels, but this may also lead to a trade-off with higher food prices and higher impact on food consumption.

In summary, measures to reduce deforestation and peatland drainage are only **effective if they are accommodated for all land using sectors, and not only the biofuel sectors.** The protection of high carbon areas reduces ILUC, but at the same time increases agricultural prices.

In general it can be concluded that the **certification of low ILUC and ILUC free** biofuels is unlikely to be able to avoid all indirect effects. **Additional measures**, beyond the scope of certification, are therefore needed, such as integrated land use planning including territorial policies.

Finally, as a comment outside the direct ILUC emissions analysis, one should take into account **economic attractiveness** of the options. As far as reduction of GHG emissions is the goal of biofuel production, it is important that the cost of reducing these GHG emissions is in line with other opportunities to reduce GHG emissions. Furthermore, if you include ILUC reducing mechanisms as reduction of food consumption in the analysis, one should include all the GHG consequences of the biofuels policy into consideration, including N_2O and CH_4 emissions from intensification, rebound effects through lower fossil fuel prices, effects on other substitutions as a consequence of increasing land scarcities like in the forestry, building and chemicals industry. And finally, in implementing a biofuels policy, the ILUC effect of biofuels is only one of the many factors that should be taken into account in an impact assessment of such policies.

4. Analysis of latest best available ILUC research

4.1. Conclusions on latest ILUC research

Based on the overview above, and the discussion on different methods in Deliverable 1, we may conclude that **economic models or the simplified representation of them** are the kernel of the best available evidence.

LCA at most uses ILUC as an extra ingredient in their analysis, where the size of the ILUC factor is derived from economic research. **Integrated assessment** models normally have an economic model as one of their components. **Historical analysis** use historical shares of area expansion in production increases as their prediction of the share of LUC accommodated through yields and therefore ignores the fact that exogenous technological is the main factor behind yield increases. **Causal–descriptive** studies take in many cases the results from economic studies with them. Therefore, our main focus is on economic studies.

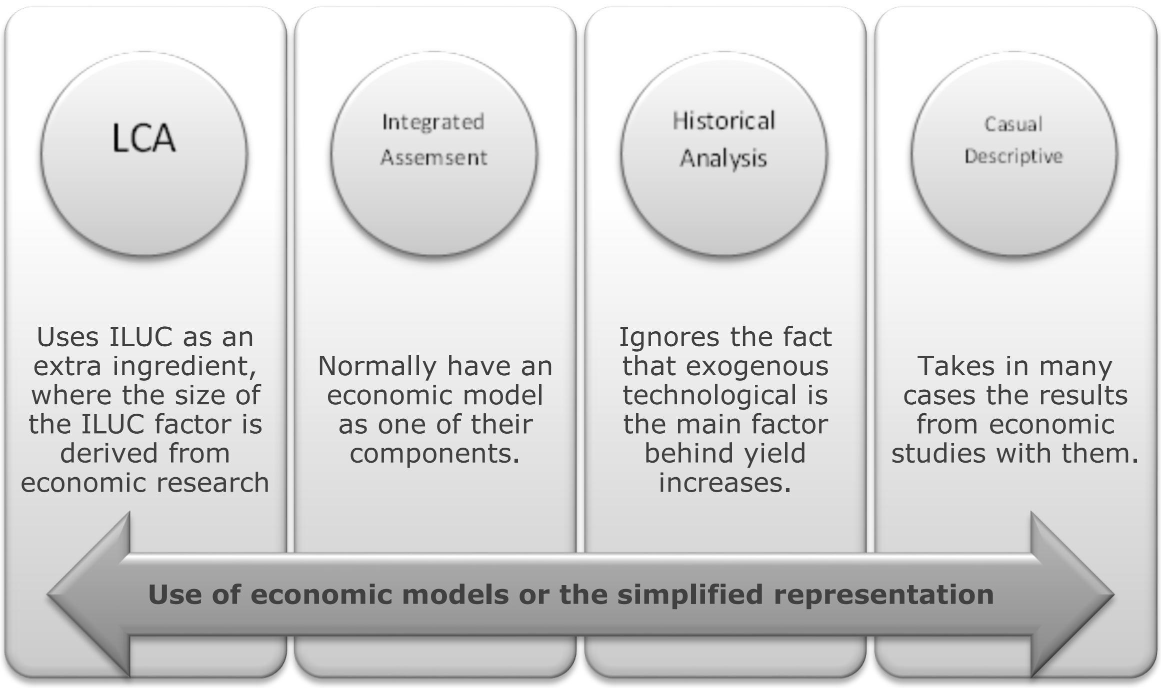


Figure 26 Approaches used to model ILUC GHG. **Source:** Own elaboration.

The most referred to studies use either the **general equilibrium model Mirage**, the general equilibrium model **GTAP** or the **partial equilibrium model GLOBIOM**. As a note, be aware that all studies evaluate land use change as a whole, not indirect land use change separately, because direct and indirect land use change are interdependent. For example, even if palm oil for biodiesel is produced on old land, this may force palm oil for food that is replaced to be produced on new and unsustainable land.

4.2. Selection methodology of best available ILUC research

To proceed with the **last selection of the literature** in order **to extract the best available ILUC** research and analyse it in detail, the following table gives an overview of the criteria we have used for the selection.

A minimum requirement is that **co-products, yield effects, land use change and emission factors** can be derived in some way, because otherwise no in-depth analysis is possible. This restricts the choice to the following post-2012 studies where as an exception we included the IFPRI MIRAGE study because it is useful as comparison with the more recent Ecofys-GLOBIOM study, and because it is much more detailed than its 2014 update. The GoVilla project is also included, even though it is not feedstock-specific, because it uses the MIRAGE modelling that is also used in the IFPRI-MIRAGE study, and it is very recent.

Table 10 Criteria used for Best Available Literature Selection. **Source:** Own elaboration.

In depth reports	The IFPRI-MIRAGE ILUC study (2011)	Searchinger on the consumption reduction in ILUC (2015)	The Ecofys-GLOBIOM analysis (2015)	ICCT's guide for the perplexed study (2014)	JRC study using the historical approach (2015)	GTAP-based analysis (2014;2016)	GoVilla project (2016)
Selected in D1	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Relevant for D1	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year of publication post-2012	No	Yes	Yes	Yes	Yes	Yes	Yes
European Research	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Information on co-products?	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Information on uncertainty?	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Information on yield increase?	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Information on land use changes per region and land use type	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Information on gCO_2/MJ	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Also analysis per feedstock	Yes	Yes	Yes	Yes	Yes	Yes	No

Therefore, we take from each type of model one or more representative studies, taking into account that for the EU the **IFPRI-MIRAGE study (Laborde 2011)** and the **Ecofys-GLOBIOM study (Valin et al. 2015)** were very important for policy. Because the IFPRI-MIRAGE study is relatively old, we take also into account its 2014 update that is mainly a type of sensitivity analysis, and the **Govilla report** that however does not have a direct focus on feedstock specific simulations results. Also a **JRC report of 2015** using historical analysis of ILUC **(Overmars et al. 2015)** is investigated because it has been influential in some way. An article by **(Searchinger et al. 2015)** in Nature suggesting that the consumption reduction as a consequence of biofuels policy should not be included as an ILUC benefit, is used as a further underpinning of the analysis of the IFPRI-MIRAGE analysis, where also results of the general equilibrium model GTAP and the partial equilibrium FAPRI-CARD model for the US are discussed. They show that the results of the other models are qualitatively in line with the MIRAGE results, although the size of the different components of ILUC differ. Another broad analysis it the **IPCC study of 2014 (Malins et al. 2014)**, that tries to give an overview of available evidence behind the models, and does an uncertainty analysis with a simplified model of ILUC derived from the more complex models.

4.3. In depth analysis of important studies

4.3.1. The IFPRI-MIRAGE ILUC study

Laborde (2011) provide an **in-depth analysis of the consequences of biofuels policy for GHG emissions as a consequence of land use change**. This implies that all direct emissions through production of feedstock, processing, distribution and consumption are excluded from the analysis. These last components will be included in LCA analyses (well to wheel analysis), and the combination of the two effects gives the total GHG effect of biofuels. Laborde et al provide total GHG savings by combining their analysis with direct saving values derived from the EC Impact Assessment.

The model

The foundation of the analysis is the **MIRAGE-BIO model**, a **global computable general equilibrium (CGE) model** using an adjusted version of the **GTAP database**. The advantage of such a model is that all sectors and regions of the global economy are modelled in a consistent manner, and that trade relations are included. The data base consists of values of production, not quantities and MIRAGE-BIO is unique in that for agricultural production and trade values are made consistent with quantities by replacing the standard values by quantities taken from the FAO database multiplied by prices (including tariffs and transport costs). Normally general equilibrium models have a small number of sectors for accounting and computational reasons, but this is partly circumvented in MIRAGE-BIO by defining more detailed biofuel-related sectors. For example, the general oilseeds sector is divided into a palm fruit, rapeseed, soybeans, sunflower and other oilseeds sector, where the oil processing sector is divided into a processing sector for each oilseed sector that produces both vegetable oil and oil meal.

The production function in standard CGE models is a Constant Elasticity of Substitution (CES) function where production factors land, capital and labour have one parameter for substitution, and all inputs have fixed coefficients. In MIRAGE-BIO this function is made hierarchical for agriculture, allowing for different elasticities between different groups of inputs, and fertilizer has been included as direct substitute for land. Parameters have been made consistent with yield and fertilizer elasticities from external sources. In standard CGE models with land use included a Constant Elasticity of Transformation (CET) function is used, the parallel of the CES function but for substitution between different land uses. Also for the CET function a hierarchy in the structure allows for different substitution elasticities for different types of land. For example, between cereals substitution may be easier than between cereals and other crops, while substitution between crops and pasture will be even more difficult. Expansion of cropland into unmanaged land is calibrated based on the literature and related with the amount of cultivable land that is available.

Crucial for a model like MIRAGE are the **values of the elasticities** used. For consumption they use elasticities from USDA. For yield they calibrate on 0.2 (p. 104) based on the CARB expert group on elasticities, and for total agricultural area they use 0.02 in most of the world (0.35 in Brazil).

Because **co-products of biofuels** are mainly used for animal feeding, it is important to model the substitution process in animal feeding. For this reason MIRAGE has developed a hierarchical CES nest structure for animal feeding. At the highest level grassland based feed substitutes with crop based feed. At the next level protein rich and calorie rich feed are substituted, while at the lowest level substitution between different types of protein respectively calorie rich feed is modelled. This allows for easy substitution between for example different types of oil cake and DDGS.

Also substitution between vegetable oils is very important for emission factors. Substitution is allowed both in feed and in food, where the elasticity of substitution is very important for the question to what extent increases in demand in one type of vegetable for biodiesel spreads towards demand for other types of vegetable oil. With perfect substitution, all increases in vegetable oil demand may go into expansion of palm oil plantations, with very large environmental consequences.

In the section below on the Ecofys study a comparison between MIRAGE and the partial equilibrium model GLOBIOM gives some further insights in characteristics of the model used.

The baseline and scenario definition

The baseline determines productivities in the reference year, and available land. They determine also the trade regime in the reference year, and the crude oil price that may influence endogenous biofuel production. It is the point of reference for the scenario's. However, the mechanisms involved in understanding the effect of biofuels policy the baseline situation is implicitly already included. Relevant for analysis is the difference between the baseline and the baseline plus some biofuels policy, where the baseline itself is of relatively minor importance.

The scenario setup is straightforward. There is basically one biofuel policy scenario where the biofuels mandate as described in the National Renewable Energy Action Plans of the 27 member states is implemented. The additional mandate is an increase from 10.2 Mtoe 27.2 Mtoe in 2020, increasing consumption by 15.5 Mtoe compared with the baseline where otherwise 11.8 Mtoe would have been consumed in 2020 (p. 37). This is consistent with a 8.6 percent blending rate, where it is assumed that the rest of the mandate will be fulfilled by non-land using biofuels. 72 percent of the mandate is biodiesel, 28 percent ethanol.

Two variants are calculated with respect to trade policy. The first one assumes that trade policy will not change. The second variant, the free trade variant, assumes full, multilateral trade liberalisation for biofuels, with contingent protection on US biodiesel remaining. Trade policy for all other commodities remains the same in the free trade scenario.

General Results

The results show that only a fraction of area increase for biofuel crops is translated into cropland change (for example for wheat and maize respectively 8 and 7%), and that 80% of the increase in crop area is accommodated through extension in managed forest and grassland (p. 11), implying that only 20% is converted from unmanaged land. This is fundamentally caused by the relative size of the elasticity of area expansion compared with the yield elasticity. The most affected regions are Brazil, Latin America, CIS and Sub Saharan Africa (p. 20, 50) and we may add South East Asia because of the large greenhouse gas consequences involved. 25% of net total cropland expansion in South East Asia is in peatland (p. 53). The GHG savings per TJ by implementing the biofuels policy are about 57% if direct emissions are calculated for, but this reduced to about 20% if indirect land use is included (p. 57, table 8), where one has to be aware that the estimated LUC GHG emissions would have been higher if the reduction of GHG caused by reduced crop (and livestock?) production would have been allocated to the reduced consumption instead of as a reduction in LUC effects of biofuels (p. 57, with a different formulation). The composition of the emissions of the biofuels emissions is roughly 35% peatland oxidation, 31% carbon in mineral soil, 30% managed forest biomass, and only 4% primary forest biomass (6% in free trade scenario) (p. 54).

Table 11 Changes in Commodity balance sheet – World – Additional mandate – No trade liberalization. **Source:** Laborde (2011)

1000 tons

	Biofuel demand	Additional Supply	Total Demand displacement	Livestock demand displacement	Ratio Additional Supply / Biofuel demand	Share of livestock demand displacement in total demand displacement
Wheat	5,366.6	-1,595.9	-6,962.5	-6,326.6	-30	90.9
Maize	4,353.0	-2,986.3	-7,339.3	-6,471.7	-69	88.2
Sugar Cane & Beet	76,616.8	69,574.6	-7,042.2	-6.6	91	0.1
Soybeans		4,677.6	4,677.6*	-1,889.9		-40.4
Sunflower		2,676.0	2,676.0*	-344.2		-12.9
Rapeseed		7,135.4	7,135.4*	-544.2		-7.6
PalmFruit		22,207.0	22,207.0*	-208		-0.9
Rice		-101.9	-101.9	418.1		-410.4
OthCrop		-765.9	-765.9	-363.4		47.5
OthOilSds		-395.4	-395.4	-322.4		81.5
VegFruits		-3,372.2	-3,372.2	25.6		-0.8
OilPalm	3,850.6	5,342.0	1,491.4		139	0.0
OilRape	4,456.9	2,474.4	-1,982.5		56	0.0
OilSoyb	2,063.5	1,270.8	-792.8		62	0.0
OilSunf	933.3	1,172.4	239.1		126	0.0
DDGSWheat		2,107.3		2,107.3		
DDGSMaize		2,261.7		2,261.7		
DDGSBeet		1,155.2		1,155.2		
MealPalm		59.8		59.8		
MealRape		3,645.6		3,645.6		
MealSoyb		5,463.4		5,463.4		
MealSunf		702.7	702.7	702.7		
OthFood		-3,139.2	-3,139.2	-114.6		3.7
Sugar		-1,881.3	-1,881.3			0.0

Source: Mirage-Biof Simulations

Note: A negative value for a demand displacement indicates a reduction in demand. A * indicates that demand displacement is positive for oilseeds since intermediate demand by the crushing sector is included here in the net demand from all sources.

The analysis of changes in demand and supply (in 1000 tons) as a consequence of the EU biofuels directive is presented in table 6 (p. 43), that is regretfully not presented per feedstock type. It shows that the increase in palm oil supply, but also sunflower oil is much more than the increase in demand for biofuels, while it is smaller for rapeseed oil and soybean oil. Overall, demand for vegetable oils for non-food purposes is reduced with about 10% of the increased demand of vegetable oils for biofuels, and roughly the same is true for non-feed and food demand for sugar cane and beet, wheat and maize (for maize about 20%). It seems that livestock demand displacement is about zero, i.e. about 16 mln tons increase in supply of DDGS and oil meals, and about the same size of reduction in demand for wheat, maize, and oilseeds. It is obvious that the price of energy content in animal feed is reduced, while the price of protein content is increased as a consequence of biofuel policies (p. 44-5) as a consequence of the increased supply of protein rich co-products. The net effect is a reduction in the total price of feed for ruminants and an increase of the feed price for other animals (table 7, p45). Land use changes are more or less consistent with this (figure 5-6 on p. 48-9). For palm oil we must be aware that demand reduction is mainly for the cosmetics industry, so not directly food related (p. 44), although replacement by fossil oil for the cosmetic industry may generate extra GHG that is not incorporated in the analysis (Searchinger et al. 2015, p. 16).

The analysis of intensification, that can be accomplished either through increase in fertilizer use or increase in use of labour and capital, is difficult, as it comprises both the effects of change in the location of production with different related yields, and the real changes in yields. The yield effects are at least positive for all commodities (p. 56, figure 13, reproduced below).

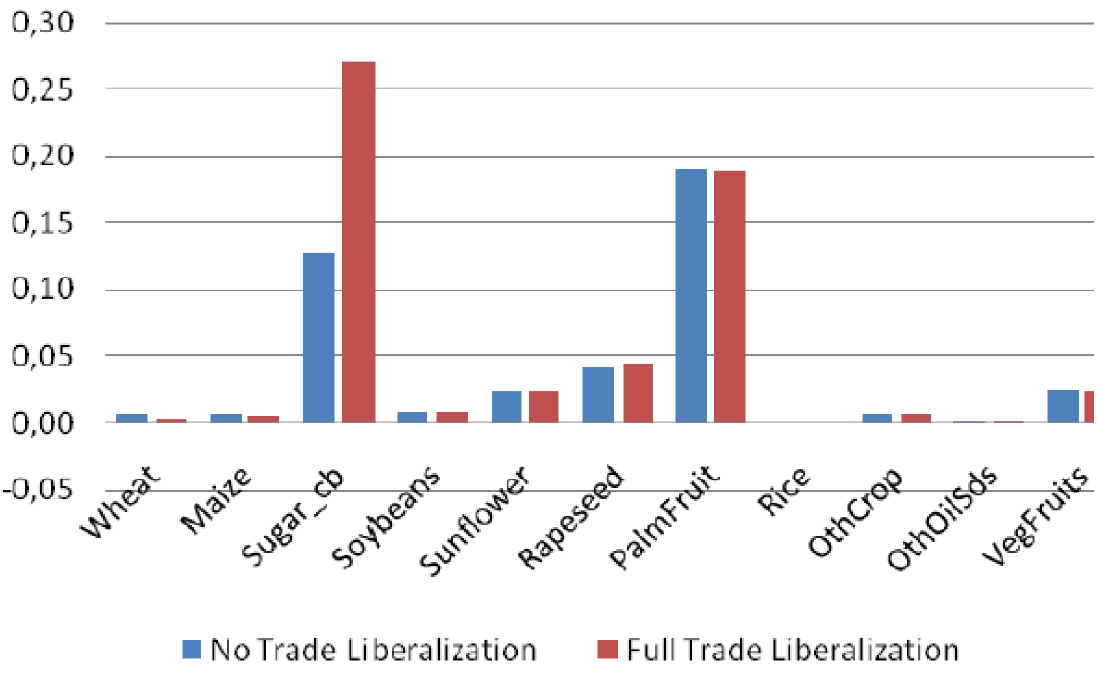


Figure 27 Effects on average World Yield. Changes compared to the baseline Tons by Ha. **Source:** Mirage-Biof Simulations

In order to investigate the effect of trade liberalisation of ethanol and biodiesel (not the other commodities) a free trade scenario has been run, with as the main effect more use of sugar cane that has higher GHG LUC emissions, but lower life cycle and total emissions per TJ (p. 46-7, 39, 58).

When production of biofuels increases, doubling of the mandate results in between 8 and 10% increase in GHG emissions (p. 17, 74), but this is mainly caused by a change in the ethanol/biodiesel mix (p. 52). The MIRAGE model is approximately linear in character in the sense that around current biofuel production levels the ILUC effects do not change a lot with the size of the biofuel production. However, there is some non-linearity in the model that may become relevant with very large changes in agricultural production. The causes of non-linearity in emissions per feedstock are (p. 63-4):

- Easiness to substitute one land for another through CET (diversification desire, differences in land quality for different crops, short versus long term perception) may explain not perfect substitution.
- CES function approach: for example in animal feeding, food processing and cosmetic industry substitution between different oils and oilcakes is possible.
- Saturation effect of fertilizers
- Below average productivity for new units of land

Crop specific results

An overview of **LUC and direct emissions per feedstock** (p. 61, table 9, reproduced below) shows that LUC emissions are relatively small for ethanol, and relatively large for biodiesel, generating net emission savings of about 50% for ethanol, and around zero for biodiesel.

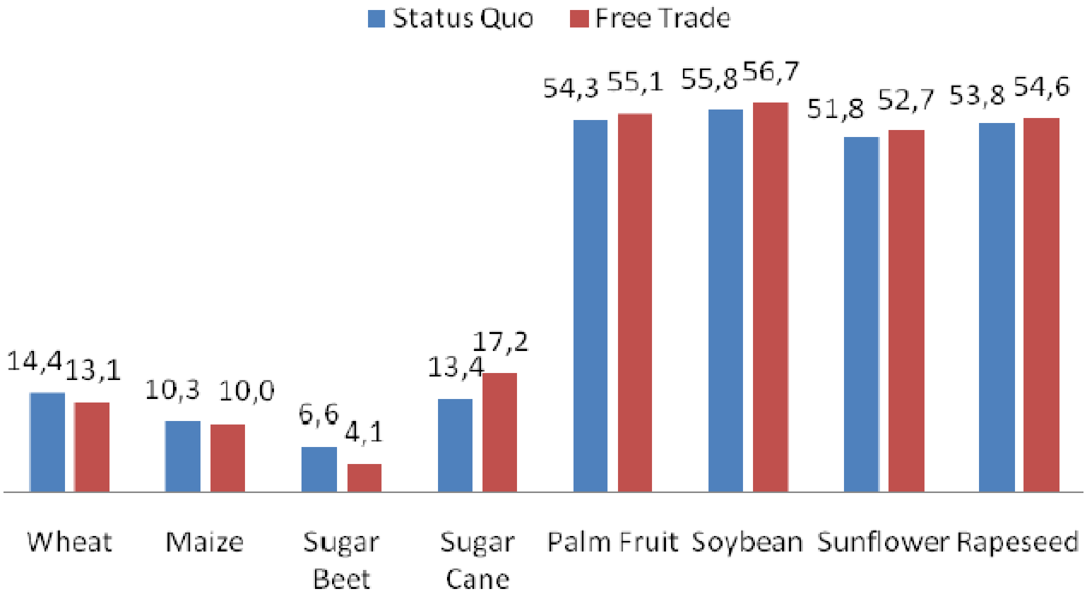


Figure 28 ILUC emission coefficients (grCO_{2eq}/MJ) by feedstock estimated at the mandate level and alternative trade policy options. **Source:** Mirage-Biof Simulations

The large effect of biodiesel is caused by the assumption that even if biodiesel production is based on rapeseed or soybean oil, in the end mainly the cheaper palm oil will increase because of easy substitution between different vegetable oils, large Armington elasticities and current trade patterns (p. 59). It is not only through the vegetable oil market, but also through the protein rich animal feed market that the mechanism works, implying that also DDGS has some effect on expansion of oil palm plantations into peatland and natural forests (p. 62). In the discussion of direct effects, we must be aware that assumptions of technology have an important role: for example, it is assumed that palm oil biodiesel is produced with methane capture facilities in order to reduce GHG emissions in production (p. 62).

Decomposition

Laborde (p. 65) explains differences in LUC by reduction in food and feed consumption, yield increases, area taken away from other crops, and area taken away from other productive uses being forestry and pasture and what is left over is expansion in pristine environments. Depending on which pristine land is involved greenhouse gas consequences are different. Cropland extension first goes into pasture and managed forest (both about 40%), then in savannah and grassland (including Brazil Cerrado) and only about 3% into primary forest (p. 51).

The interpretation of the replacement ratio presented in table 10 (p. 66) is complicated, because it involves many processes: substitution with other crops because of price increases, competition with land, and by-products.

With respect to the consequences of by-products and land competition for animal feeding, table 11 (p. 67) shows that the increase in DDGS supply because of maize or wheat ethanol production replaces mainly maize and wheat in animal feeding, and only a little bit vegetable oil meals, where in tons total animal feed supply is increased. The increase in oil meals because of biodiesel production is also substituted by a reduction in maize and wheat, but also of soybeans in animal feeding. We have to be aware that the increase in oil meals is smaller than the by-products produced by biodiesel production because also the oil meal market reacts to the extra supply.

Also table 12 (p. 68) on the decomposition of production change in area expansion, yield increases because of fertilizer use and yield increases because of increase of non-and production factors, is not easy to interpret because it combines yield changes as a consequence of intensification with yield changes as a consequence of relocation of production in regions with lower or higher yields (p. 67). In the table everything is attributed to either changes in factor or fertilizer use (becoming sometimes even negative because of international relocation) where part will probably be linked with differences in yields in different regions. The basic conclusion of this table is that about 80% of production increase is accommodated through increase in land use. However, we must be aware that this is about the increase in crop production of specific feedstock crops, where through intensification of other crops and livestock more land can is freed.

Table 14 (p. 70) and appendix III (p. 95-6) show per feedstock land use changes per region and land use type. This allows for an interesting decomposition of land displacement for different biofuel feedstocks. For example, for biodiesel from palm oil it shows that this not only leads to more palm fruit production in south east Asia and Sub-Saharan Africa, but also to increased soybean and sunflower production (which seems surprising because table 10 on p. 66 suggests that additional supply is only 3.4% lower than biofuel demand from palm oil). Because of yield changes and consumption reduction the net increase in cropland is only 1/3 of the increase in energy crops (which must be 3.4% lower than crops used for biodiesel, according to table 10), and 46% of this increase in cropland is accommodated through a decrease in pasture land and 48% through a decrease in commercial forest land, implying only a reduction in pristine land of 2% of the increase in area for energy crops.

With respect to sugar cane ethanol, the analysis is more or less the same: 1/3 of expansion of sugar cane area is translated into expansion of cropland area, 60% of this increase in cropland area is at the cost of pasture area, and 30% at the cost of commercial forest, leaving only 10% for expansion into pristine area, being 3% of the increase in sugar cane area.

For maize the effect is even stronger. An increase of 6.52 ha maize/TJ results in an increase in an overall cropland expansion of 0.88 ha, i.e. 13% of the feedstock expansion, where we have to be aware that according to table 11 (p. 67) (on the evaluation per feedstock scenario of changes in wheat, maize, rapeseed and soybean) that the increase in wheat area is less that the area needed for the feedstock, partly because of co-products, and partly because of price increases. Half of this crop area increase is accommodated through livestock area and the other half through commercial forest, implying no consequences for pristine areas. Because the prices of feed for ruminants are reduced a little bit in most regions (p. 45) part of the intensification may be accomplished through extra feed from crops. Figure S4 of Searchinger et al (2015), reproduced below, gives a good insight into the dynamics of maize on LUC.

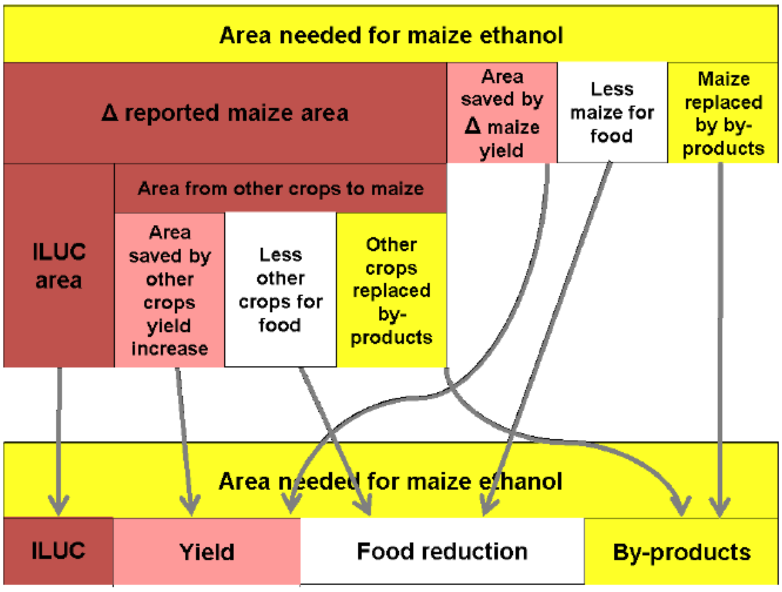


Figure 29 Conceptual Decomposition of the Land Area Results of Additional Corn Ethanol Production in IFPRI Model. **Source:** Searchinger et al (2015).

Searchinger et al make the decomposition of the IFPRI results by starting with the total area needed for the maize feedstock, i.e. 14 ha/TJ. They estimate that by-products reduce this area use to about 8 ha/TJ by calculating the land use of the feed that is replaced. Of this area almost 50% is reduced because of reduce food consumption, and of the 4 ha/TJ that is left over almost 70% is accommodated through yield increases. The analysis of wheat ethanol is more or less the same[4], where the analysis on p. 96 of the IFPRI-study shows that from the cropland expansion 40% is accommodated through pasture area reduction and 7% through pristine areas.

If land use changes are known, then applying carbon stock values generates the GHG emissions from land use change. Some of the carbon stocks are presented in appendix II (p. 93-4) showing that primary forests have a higher carbon stock than commercial forests, but not showing the carbon stock in livestock, although this is probably included in the soil carbon stock.

So, although the table gives a lot of information on the enormous reduction of original expansion of area through yield increases and consumption change, the additional analysis as made by Searchinger et al was necessary in order to split the land use changes in consumption and yield changes.

The analysis of the results shows that reduction of demand for feed and food plays a critical role (p. 16) and Searchinger et al make this explicit for crop demand. On the other hand, it may be that also meat consumption is influenced by the change in feed prices and the reduction in area for livestock. This is neither discussed by Laborde et al, nor by Searchinger et al.

Uncertainty and sensitivity analysis

Laborde et al distinguish the following **sources of uncertainty** (p. 24-27), where we have combined some categories in a little bit larger groups:

- Crop yields in baseline; relatively small;

[4] The same analysis for wheat is also published by Laborde et al (2014) based on an analysis by JRC, with a little bit different results as Searchinger, but with the same message.

        - marginal yields may be lower or higher
    - Crop yield response versus area response (point 8 and 2 together)
        - Includes response on fertilizer
        - Includes both substitution between different land uses (including commercial forest (p. 20) and agricultural land expansion into pristine areas.
    - Demand response for all crops
    - Substitution possibilities between vegetable oils
    - Livestock sector:
        - Intensification possibility
        - Flexibility feed ration
        - Demand response of meat
    - Average carbon stock per ha; does use of average carbon stocks bias the results?
        - Peatland emissions
        - Agronomic practices, like depth of tillage, use of genetically modified soybeans that may reduce need for tillage, etc.
    - Trade and business networks, trade policies, exchange rates, may change competitive positions of different regions.
    - Land governance: may reduce expansion in high GHG areas.
    - Public investment in infrastructure
    - Public R&D
    - (Oil prices: if they are high enough to make biofuel production in some regions profitable, biofuel price increases may reduce biofuel production.) Also lower crude oil price may generate extra demand.
    - For net emission balance of biofuels is also relevant:
        - Processing pathways, source of energy used for processing, capacity to innovate in this and to do investments to make new technologies effective.

A Monte Carlo sensitivity analysis is accomplished (p. 15) where the distribution of parameters can only be set by assumption, not econometrics (p. 74), where basically all points should be considered, not only the average results. The parameters that are shifted are the share of extension into primary forest, the price elasticity of intermediate demand for agricultural products, the ratio between average and marginal yield, the elasticity of substitution between land and other production factors, the elasticity of substitution between land and feed/fertilizer, the elasticity of transformation of land, and the land extension elasticity (p. 30). The conclusion is that most uncertainty affecting ILUC is outside the EU, because in the EU relatively little intensification possible (p. 19).

Be aware that a limited range of parameters is analysed, and that household behaviour, substitution among subsets of inputs, differences in LCA savings and uncertainty in carbon stocks is not investigated (p. 33, 75).

In Laborde et al. (2014) a further sensitivity analysis is accomplished with some parameter changes, but although this changes the GHG effects a little bit, the basic message remains the same.

Laborde et al did also some analysis on changing the closure, i.e. by restricting adjustment through consumption and feed substitution:

> "The previous section focused on a Monte Carlo analysis focusing on key parameters, mainly on the supply side of the model. As discussed in the first part of this chapter, we have seen that demand displacement plays a critical role in explaining the low LUC effects of key crops, in particular corn and wheat. Figure 23 looks at this issue by presenting the LUC emission coefficients assuming alternative closure on the food and feed markets. Compared to the standard closure where food and feed demand can react freely to prices, we investigate two alternatives: fixed food consumption by households and no substitution between crops and co-products. However, these approaches still allow for food adjustment. When food consumption is blocked, the intermediate consumption mix of the processing sectors can still evolve (replacement of vegetal fats by animal fats or decrease of the average contents of flour in processed food, etc.); similarly, when the crops-co-products substitution is restricted, the overall level of meat production can still adapt." (p. 82)

Interpretation

Laborde et al (2011) give a number of thoughts with respect to the interpretation of the results. First, they broaden the perspective of LUC emission analysis. GHG emissions are not the only relevant aspect of biofuel policies. Biofuel policies may also be designed to achieve other goals like energy diversification and farm support (p. 17). Second, analysis of LUC emission is almost always about biofuels, but such analysis should also be done for other policies, like CAP reform and trade negotiations. A discriminatory approach to ILUC for biofuels is inefficient and potentially unsustainable both politically and legally (WTO) (p. 17). For example (p. 86), Laborde (2010) shows that DOHA may have larger effect than biofuels.

Second, they warn against the use of crop specific LUC coefficients, because a differentiated policy based on LUC per crop may be difficult because of risk of leakages (p. 18). A higher direct saving as goal is better, where an average ILUC is taken into account (see also p18, 86-87). They promote a higher ethanol share, because it has significantly lower ILUC.

Third, alternative trade policy options may be developed:

> "if liberalizing the ethanol trade will help to reduce net emissions, even if land use effects can be contrasted, alternative trade policy approaches can be put on the table. Indeed, facing the risk of additional imports of feedstocks from countries with weak land use governance, the EC may want to enforce incentives through trade barriers/preferences. The measures should not be "unfairly" discriminatory and should avoid extra transaction costs, including certification costs at a firm level. The idea is to promote good governance of land resources, tackle externalities, and promote adhesion to ongoing international initiatives such as the United Nations programme on Reducing Emissions from Deforestation and Forest Degradation (UN-REDD). The GSP+ reform, linking trade preferences to implementation of international conventions (e.g. Kyoto, biodiversity) is such an example. In this context, EC trade policy should be used to provide incentives to prevent large leakage effects associated with narrow sustainability criteria. In addition, MFN based treatment could be rethought. Instead of strengthening sustainability criteria/certification for biofuels and for the feedstocks to produce them, all imports of the relevant products, whatever their uses, could be covered by the new discipline. In order to avoid restraining existing market access on EC partners and risking a WTO dispute, but also hurting existing importers, the EC could provide tariff rate quotas equal to the current level of imports that will not need to be certified, or equivalently, a number of "free" certificates based on a kind of "grandfathering" principle." (p. 87)

Fourth, using technologies to increase yield (biotech) and low carbon agricultural practices may be important to mitigate emissions by reducing additional land requirement (p. 18, 87), where increase of R&D cannot be expected to have returns in the short term.

Fifth, we have to be aware that emissions from additional fertilizers because of intensification are excluded (p. 13), although it is suggested that these effects will be relatively small (p. 110). Leakage of GHG emissions through changes in crude oil prices (estimated at 0.94%, see p. 57) and petroleum use, are also excluded, where it is suggested that the reduction in crude oil price of 0.94% generates an increase in demand for oil implying that the reduction in crude oil demand is only 70% of the amount of fuel produced by the biofuel. Also electricity as by-product is not accounted for (p. 13), because it has no ILUC effects. However, because it has direct GHG implications, it is included in the LCA and therefore in the net emission calculation.

Sixth, because of the lack of flexibility in mandates the fixed amount of extra biofuel demand will increase price volatility on agricultural markets with unintended consequences for poor consumers: to prevent this, more flexibility should be put into the biofuels policy (p. 88).

Finally, because of all uncertainties, they suggest that flexibility in policy and a regular health check is necessary in order to incorporate new insights into policies (p. 18).

(Laborde et al. 2014) provide some further comments on the results. They mention that the yield elasticity estimates should be improved (p. 28), and that crop yields are in general on the high side. Substitution between vegetable oils on value instead of quantity terms perhaps would be better. The rate of technical improvement of yields may also depend on crop price, but if the effect is there, it will require a long period of time before it has effect (p. 28-9). Finally, they comment that taking the value of wood into account would increase deforestation in the baseline requiring less deforestation for biofuels (p. 29).

4.3.2. Searchinger on consumption reduction in ILUC

Searchinger et al (2015)[5] argue that according to the three most important models used in the context of biofuel analysis for government policies in Europe and the United States (GTAP, FAPRI-CARD and MIRAGE) rely on food consumption reduction in order to generate greenhouse savings for biofuels (p. 1420). Their starting point is the idea that the combustion of biofuels just as the combustion of fossil fuels emits greenhouse gasses, and that it has to be proven that there is an offset in emissions by land use change (p. 1421). Biofuels may be produced by using land that may expand into forests and grasslands generating greenhouse gas emissions. This expansion may be reduced by replacing crops for food, or by increasing yields on cropland. As far as it is at the cost of crops for food, it will imply that global food consumption decreases and this will be disproportionally be the case for the poor. Regretfully, this share of LUC reduction generated by food crop reduction is not analysed explicitly in most reports.

By making the effects explicit, it becomes clear that 23 to 53 gCO_2eq per MJ wheat or maize ethanol is reduced because of diverted food, leading to a reduction in GHG because people and animals will respire or waste less food (p. 1420). If this offset of carbon emissions from reduced consumption would not have happened in all analyzed models the use of wheat and maize ethanol as biofuel would increase GHG emissions compared with the use of fossil fuels (p. 1421).

The size of the consumption effect depends on the supply and demand elasticities (Plevin et al, 2015; Roberts and Schlenker, 2013). We have to be aware that next to food quantity also food quality is reduced (p. 14).

Next to the reduction in food production also the increase in yields has an important role in the models analysed, with an effect in the order of magnitude of the food consumption reduction (p 1421). In a model like MIRAGE the increase in production is mainly accommodated through yield increases, while the literature suggests that the area response is much higher than the yield response (Berry, 2011). History shows a

[5] The pages above 1400 are the article, the low page numbers the pages in the supplementary materials.

doubling of yields between 1961 and 2015 despite a reduction in prices.[6] By the way, Searchinger et al calculate the yield as the average yield of where area change occurs, so including yield changes because of area shifting over the world (p. 19).

Let us dig a little bit deeper into the yield analysis (p. 23). The main issue is that yields increase over time because of changes in technology, so yield increases are at least partly not related with higher prices or demand increases. General improvements in the quality of seeds, machinery and chemicals will increase yields, as does infrastructure development such as roads. Increased education will increase farmer performance as well, and investment in drainage or irrigation also happened. Government has invested significant amounts in R&D that is partly focused on yield improvements. So, even though increases in crop production were for 83% accommodated by yield increases between 1961 and 2005 this does not imply that yield increases were caused by these production increases. The fact that real crop prices fell with about 50% during this period suggests that the yield increases (and other cost reduction because of technological change) caused the price reduction, and not the other way round. This is consistent with econometric evidence (Huang and Khanne, 2010; Berry and Schlenker, 2011; Goodwin et al, 2012). Berry (2011) suggests that the literature review by Keeney & Hertel (2009) is more correctly seen as suggesting no yield response than the rather higher yield response they take from it.

An important issue is the endogeneity problem (p. 25), because yield reacts to prices as do prices react to yields. It is exceptional that this problem is tackled in the econometric estimations and where it is, the indication is that yield elasticities are not significantly different from zero. And most studies suggest that yield elasticities are at least much lower than area elasticities, while MIRAGE does the contrary, explaining their large effect on yields. By the way, Searchinger et al refer to table 12 (p. 68) of Laborde et al (2011) where the yield effect seems to be small (nicely decomposed into a fertilizer and factor increase effect!) but that this suggestion is wrong because this table is on yield changes in biofuel feedstock production where yield increases are

[6] This makes it only plausible, because yield increases is one of the ways to reduce cost.

especially high in other crops than the feedstock (p. 27)[7]. As a consequence Laborde et al predict that only 12-13% of net feedstock expansion is accommodated through land expansion (p. 28).[8]

Searchinger et al do their basic calculations based on carbon content (although some calculations are made in areas, p 17), so the food quantity reduction is defined as the reduction in carbon in crops available for food and consumption as a consequence of ethanol production (p. 5). Their decomposition method is as follows. First gross feedstock demand is calculated. Second, by-product production is determined, based on carbon content. Third, crop replacement related with these by-product is calculated. Fourth, consumption reduction is calculated. Then the area calculation is made, i.e. gross area minus area reduction because of by-products, and area reduction because of reduced consumption. Finally, the area is further reduced by an increase in yield (p. 1421).

The basic conclusion is that models should make at least explicit what the trade-off between biofuels and food is, so they could use estimates of LUC emissions without taking the consumption reduction into consideration (p. 1422).

4.3.3. The Ecofys-GLOBIOM analysis

The model[9]

GLOBIOM is a global partial equilibrium model, implying that only the relevant sectors of the economy are modelled, i.e. agriculture, animal feed, forestry, and biofuels. The model is an optimisation model where producer plus consumer surplus is maximized given a number of explicitly described techniques where the inputs used are exactly described. Where in MIRAGE substitution between inputs for production is modelled in a general way by one function, in GLOBIOM substitution between inputs is modelled by choosing between explicitly described management techniques. In order to prevent that the model jumps from one technique to another too fast, some adjustment costs are normally introduced. The choice implies also that the number of

[7] As a consequence of the low CET elasticities at more aggregated types of land use.

[8] Another note is that Laborde et al don't take into account that as a consequence of the biofuels directive the production of biofuels elsewhere is reduced a little bit (p. 30). It is not fundamental for the outcomes.

[9] This section is mainly based on Annex I of Valin (2015).

predefined techniques determines what can be chosen in the model, while in a general equilibrium model the general idea of substitution is introduced as an implementation of the idea that in reality many more substitutes are available than where we can think of in advance.

The **database** of GLOBIOM has mainly physical quantities combined with explicit prices, where in MIRAGE it is on a social accounting matrix filled with values, where only analytically a difference is made between volumes and prices and where the quantity balances in trade, production and consumption are not guaranteed. To solve the last problem partly, in MIRAGE the database in values has been recalibrated consistent with quantities and prices from FAO.

In GLOBIOM land use is modelled at **grid-cell level**, while in MIRAGE it is modelled at a regional level, although split over 18 agro-ecological zones. As a consequence land conversion possibilities in GLOBIOM can also be modelled at a grid-cell level, taking into accounting suitability and protected areas specifically, while MIRAGE can only allocate land based on availability and general elasticities.

With respect to other production factors a model like GLOBIOM has opportunities to take water availability and irrigation techniques into account on a grid-cell level, while a model like MIRAGE can only do this at a national level, and is in practice not doing anything with water. On the other hand, a model like MIRAGE can take dynamics of the labour and capital market into account, where normally partial equilibrium models have only a general cost function. The problem with the last is that it is difficult to model comparative advantages, although you may doubt to what extent this is done realistically in general equilibrium models.

A general equilibrium model like MIRAGE tends to have less sectors, but the sum of all sectors equals the total economy. In GLOBIOM more agricultural sectors are described, but sectors for which no technological descriptions are available are excluded (about 16% of harvested area). In MIRAGE the most relevant sectors have been added to the database to solve the problem of the lack in detail.

In GLOBIOM yields are grid-specific, based on a crop growth model that is scaled towards aggregate FAO yields. Global harvested area in FAO-statistics is 78% of arable and permanent cropland, implying that a lot of land is not harvested even once a year because it is used as temporary meadows, or is idle or abandoned land. Both GLOBIOM and MIRAGE assume normally that this fraction will not change over time, but in some cases abandoned or idle land can be taken into production for biofuels, or double cropping is implemented.

The optimization used in most partial equilibrium models like GLOBIOM allows for modelling feed composition explicitly, while a model like MIRAGE uses a general function to model this substitution, providing less opportunities to include explicit restrictions on feed rations like the amount of phosphorus included in the feed.

Both MIRAGE and GLOBIOM model technological changes partly as exogenous based on breeding, new seeds, and new technologies, and partly as a reaction on prices. GLOBIOM reacts on price changes by explicitly choosing other management systems, while MIRAGE has a general function where fertilizer and land can be substituted, and where labour and capital intensity may change.

Because of technical limitations, consumption modelling also differs between the models. In GLOBIOM each commodity has a separate demand function, while in MIRAGE all consumption is modelled through one system of equations, allowing for substitution between different consumption goods (p. 128). The lack of substitution possibilities in GLOBIOM restricts some relevant interaction possibilities in consumption. It is by the way not necessary to have such a simple consumption function in all partial equilibrium models; for example CAPRI has a much more developed demand system.

Trade is handled differently in GLOBIOM compared with MIRAGE (p. 117-8). In GLOBIOM trade is in quantities where goods are perfect substitutes. In MIRAGE trade is in values (implying that accounting in quantities is not automatically consistent and with the trade functions used will not become inconsistent after trade) and commodities from different regions are imperfect substitutes. Furthermore, trade in MIRAGE depends on current trade, implying that it is difficult to generate new trading regions in the model or expand the exports of regions with small exports, while GLOBIOM automatically allocates trade to the cheapest suppliers, including frictions like transport cost and tariffs. The approach in GLOBIOM requires a proper modelling of production cost, because otherwise exports may be allocated to the wrong regions.

With respect to greenhouse gas emissions both models can do the same job (by a separate model coupled to the main economic model). Peatland oxidation, soil organic carbon, and natural vegetation conversion emissions are calculated in both models, while natural vegetation reversion and agricultural biomass is only taken into account in the Ecofys study. Neither of the studies includes agricultural production and chain emissions (direct and indirect) like emissions because of cultivation of crops (fertilizer production and use), machinery, conversion into biofuels, product transport and distribution[10], because they are part of direct emissions and included in an LCA analysis. Neither of the studies include non-CO_2 emissions because of intensification in agriculture.

The baseline and scenario definition

The baseline determines productivities in the reference year, and available land. It determines also the trade regime in the reference year, and the crude oil price that may influence endogenous biofuel production. The baseline may be relevant to become aware that because of increase in livestock, fertilizer use, rice cultivation and other crop expansion agricultural emissions increase in GLOBIOM between 2010 to 2030 from 2710 MtCO_2 to 4440 MtCO_2 (p. 25), and that crop prices have a downward trend because of productivity increases. It shows also what land is potentially available for biofuels production, i.e. if agricultural land area is reducing this may be taken into production for biofuels with relatively low carbon cost.

Effects of multi-cropping are included in yield changes (p. 21). Yields are expected to rise with 1% per year on average, and livestock feed conversion efficiencies to increase by 30-50% between 2010 and 2030 (p. 19). The baseline is the point of reference for the scenario's, but the baseline is not crucial in understanding the results from the scenarios compared with the baseline, and therefore will not be discussed here further.

The basic methodology is a baseline run and to compare this with a scenario that is equal to the baseline except for increased biofuel production. For specific policies like deforestation restrictions, restrictions on expansion of palm oil plantations into peat

[10] p. 4. Here it is mentioned that also emissions from land use management are included in the model; but these results are not included in the report further on.

land, use of abandoned land, specific model assumptions have to be made. The mix of second generation is determined by the model, while the mix of first generation is determined by the National Renewable Energy Action Plans (NREAPs) (p. vii, 31), but because of lack of data some feedstock shares are based on USDA estimates for 2013 that are kept constant (p. 32). 16% of this mix is palm oil, which has very large GHG effects because of peatland oxidation.

Scenarios are assessed for the year 2020. There are scenarios that analyse the consequence of a small increase in biofuels from a specific crop, scenarios where groups of crops are shocked together, and scenarios on the EU 2020 biofuel mix (p. viii). In the last case some attempts to make biofuels more sustainable are analysed, like a 7% cap on EU biofuels, more use of abandoned land in the EU, limiting deforestation by putting a price on deforestation, and putting a ban on peatland. Most biofuels, including second generation biofuels, are included in the analysis, but for example biofuel from cooking oil and fat is not analysed.

General results

In the EU cropland expansion is mainly converted from abandoned land (p. 46), where outside the EU forest in Southeast Asia comes first, followed by other natural land and grassland in Latin America, and other natural land and grassland in Southeast Asia. The other regions have relatively minor changes. For the EU biofuel mix also Sub Saharan Africa natural land seems relevant (p. 84).

The EU 2020 biofuel scenario shows that 8 Mha of land is converted into cropland and 0.8 Mha into short rotation plantations (P ix, 85). Of the 8.8 Mha, 2.9 Mha takes place in the EU through a reduction in the area of abandoned land, and 2.1 Mha is converted in Southeast Asia as a consequence of expansion of oil palm plantations, half of which is at the cost of peatland and tropical forest. Especially for biodiesel drainage of peatlands is an important and long-lasting source of greenhouse gas emissions (P. x, 40). Advanced biofuels have negative LUC emissions, partly because of more carbon sequestration because of no-tillage practices for most of these feedstocks. Forest residues have relatively high soil organic carbon emissions that are counted even though it is not because of LUC. For straw that is unsustainably harvested the GHG emissions are 16 g CO_{2e}/MJ because of reduced carbon sequestration and reduced productivity, but when it is only sustainably harvested (once in two or three years) the emissions are reduced to zero.

An essential mechanism is the effect of the increase of protein meals as a by-product of most biofuels: although part is accommodated through substitution in the animal feed ration, part is accommodated through reduced protein meal supply, implying that depending on the substitution possibilities between vegetable oils a part of the extra supply of biofuels is accommodated through the low by-product and cheap vegetable oil from palm trees, generating GHG emissions through peatland oxidation (p. 40). The increase in the supply of protein meals of 13 Mt boosts milk (1Mt) and meat production (0.2Mt), and the extra demand for vegetable oils for biofuels reduces other demand for vegetable oils with 2.4 Mt (p. 85).

Introduction of a cap of 7% of biofuel from conventional biofuels implies an increase in the share of advanced biofuels, but because they are counted double or more, the total share of biofuels in transport fuels is also reduced. Both effects generate a reduction in GHG emissions with such a policy (p. 41).

An effective ban on peatland drainage combined with a very low deforestation scenario by putting a price of $50 on deforestation may reduce ILUC to about 5 g CO_2eq/MJ. This obviously requires that the policies are effectively enforced (p. 42). The consequences for food prices are not discussed.[11]

Decomposition of crop specific results

In order to get a grasp on the dynamics of the model and because a lot of information on the specific feedstocks is given in the report, we try to interpret a small **number of feedstocks**, where in the next section the decomposition is integrated in a general overview. In order to get a good grasp of the dynamics of LUC for specific biofuels, we try to interpret text and figures in the report (p. 49-50). It requires a lot of recalculations, because some numbers are in GH/ha, other ins ha/TJ, and a lot is per 123 PJ. The scenario's on second generation biofuels and the mitigation scenarios discussed in separate sections.

Ethanol from wheat

For ethanol from wheat, energy productivity in the EU28 is 23.8 ha/TJ, composed of 130 ton wheat/TJ ethanol and a land productivity of 5.46 ton/ha; the last is more or less consistent with the 4.96 ton/ha (7.2/1.5=4.8 ton/ha for the EU when calculated

[11] Although some people argue that for example in Brazil intensification of livestock is very easy to accomplish, and so if the ban on deforestation is effective, ILUC GHG can become even negative

based on the additional feedstock production, but there a yield effect is included) that can be derived from the numbers on location of production and land use of additional feedstock production.

Table 12 Attempt to interpret area changes for wheat ethanol, per TJ**. Source:** Own elaboration.

	t/TJ	ha/TJ	%
Gross feedstock area	5.0	23.8	
Co-product	1.3	6.2	26%
Net feedstock area	3.7	17.6	
Cons reduction	1.3	6.2	35%
Area without yield change	2.4	11.4	
Yield effect		3.2	28%
Calculated crop area expansion	8.2		
Reallocation effect		-4.6	-57%
Cropland effect		12.8	
Grassland reduction		2.4	19%
Agricultural land		10.4	

The first step is co-product produced, that is in tons 26% of the feedstock. The next step is a reduction in feed and to a lesser extent food demand that is caused by increased wheat and other agricultural prices, being 26% of the extra production of wheat for biofuel. The small effect (3% of wheat production) on consumption is caused by low consumption elasticities for cereals in the EU (around -0.1), where feed demand seems to be much more elastic. However, the effect seems large given that also the area of grassland is reduced, implying that meat production must have been reduced about which nothing is mentioned in the text.

We should mention, that according to the calculation above wheat production is increased by 58% of the area needed for ethanol production, while in practice it is 76%. The reason may be that wheat DDGS is not substituting with wheat, but also with maize and oilseeds.

The increase in maize production is for 17% accommodated through yield increases, but the increase in crop production according to the graph for about 30%. This would generate an increase in demand for cropland of only 9.1 ha/TJ, while according to the text 12.8 ha/TJ is converted in practice. This difference of 40% of our calculated crop area we call the reallocation effect.

What is the cause of the reallocation effect? First, only 57.5% of wheat is produced in the EU, so productivity may be lower than in the numbers of the energy productivity that is based on EU28 average yields (that is different also for the EU if you compare. However, the yields according to the data in the section on additional feedstock production show that the yields are more or less the same. Second, the substitution of DDGS with oilcake may give a difference because soybean yields per ha are much lower than maize yields[12].

From the difference between cropland and agricultural land displacement it may be concluded that 12.8-10.4=2.4 ha/TJ is the reduction of livestock area (that may be explained by animal feed becoming cheaper, but this seems to be inconsistent with the decrease in feed demand referred to in the adjustment to the shock. So, it must be explained by land competition, where the main reduction in grassland are according to the graph in North and Latin America.

If we look at the GHG emissions graph, it becomes obvious that soil organic carbon from converted natural land is the main cause of GHG emissions. The small increase in drained peatland for palm oil that compensates for the reduced need to produce animal feed from other feedstock, has a relatively large impact on GHG emissions when compared with the area involved. Forest reversion in the EU is the other important component, as is the soil carbon loss of conversion of natural land in the EU. In table 13 we have tried to relate the carbon stock change explicitly to the land use change.

[12] According to p. 210, corn yields are 8.4, where soybean yields are 2.6.

Table 13 Area and Carbon stock change as a consequence of an increase of wheat ethanol demand. **Source:** Own elaboration.

	per 123 PJ		Per TJ	
	Area	$MtCO_2$	Area	gCO_2/year
nat land EU	-750	-54	-6.1	-22.0
abandoned	-490	-29	-4.0	-11.8
peatland	-34	-11	-0.3	-4.5
cropland	1600	15	12.8	6.1
grassland	-260	-4	-2.2	-1.6
nat land FSU	-100	-1.5	-0.6	-0.6
		-84.5		-34

After having finished this attempt to interpret the results, some puzzles remain. It is mentioned that total demand for cereals (assuming this exclusive wheat) is reduced by 3.6 Mt, so this is 60% of the demand reduction for animal feed.

In summary, it is difficult to get a grasp on the precise dynamics in the model. First everything has to be scaled to one numerator, but then it is difficult to get into a consistent story. One of the puzzles is why demand for feed is reduced next to a reduction in grassland. This would imply a large reduction in livestock production, about which nothing is mentioned. The difference between area effects with quantity effects may be explained by differences in land productivity for different crops used for animal feeding.

Calculation LUC by Laborde (2011), mainly based on Searchinger (2015)[13]. **Source:** Own elaboration

	ha/TJ	
Gross feedstock area	20.2	
Co-product	7.8	39%
Net feedstock area	12.2	
Cons reduction	5.9	49%
Area without yield change	6.2	
Yield effect	4.8	77%
Crop area expansion	1.4	
Grassland reduction	0.5	35%
Agricultural land	0.9	

If we compare this analysis with the IFPRI study, it seems that the effect of by-products is a little bit smaller, that in the IFPRI-study mainly food demand is reduced while in the Ecofys study it is mainly feed demand, and also the yield effect is much smaller, with as the end result that cropland expansion is much more 12.8 ha instead of 1.4 ha. Also the percentage effect of crop area expansion on livestock area is smaller, although the absolute effect is larger. A problem in the comparison is the reallocation effects that increase cropland use, but even if we would use the lower final effects, the qualitative difference would be the same.

Table 14 An attempt to calculate GHG emissions per ha from IFPRI (2011). **Source:** Own elaboration.

	Carbon stock	Area	GHG emissions
	tCO_2/ha	ha	tCO_2/year
Carbon from mineral soil	100	0.5	2.7
Carbon from commercial forest	250	0.8	9.4
Carbon from natural forest	300	0.1	1.5
Peatland	1100	0.06	3.3
Total		1.39	16.9

[13] Laborde et al (2014), p. 11 provides a little bit different calculation, with the same message

Despite that cropland expansion is a factor 10 higher in the Ecofys analysis, the difference in GHG emissions is only a factor 2 (17 g CO_2 eq/MJ instead of 34). We tried also to give an impression of the type of calculation that may be behind the IFPRI-study in table 15, where it is assumed that 30% of palm oil expansion is into peatland, and that all other conversion have factors at the high end of tables A3-A4 (p. 93-94) and indications of land use changes is taken from table A8 (p. 96). So, the dynamics behind the GHG emissions from wheat ethanol is completely different between the IFPRI and Ecofys studies.

Ethanol from maize

Production analysis

The first step is the analysis of the effect of maize ethanol production on the maize market. Energy productivity of ethanol is 64 GJ/ha, implying that 15.6 ha/TJ, or 1.9 Mha per 123 PJ is needed. The production of 123 PJ ethanol requires 14.2 MT of corn, implying 115.4 ton/TJ. The implicit yield is therefore 14.2/1.9 MT/Mha= 7.5 ton/ha. Because 18% of ethanol production is outside the EU, total land use and yields for maize may be a little bit different from the those calculated above.

The shock of 14.2 MT maize is not the final shock of maize production, because 26% of 14.2 MT = 3.7MT of DDGS is produced as a co-product that is used as a substitute for protein animal meals. We have to be aware that in this case tons of DDGS are handled as if it were tons of maize, where in practice DDGS will only partly substitute for maize, but also for other grains and oil meals.

Table 15 Production and land use effects of maize ethanol per TJ of ethanol. **Source:** Own elaboration.

	Ton	ha	%
Gross feedstock area	115.4	15.6	
Co-product	30.0	4.1	26%
Net feedstock	85.4	11.5	
Feed demand	24.8	3.4	29%
Production increase	60.6	8.2	
Yield increase		2.3	28%
Cropland change (calculated)	5.9		
Reallocation effect		-1.8	-30%
Cropland change		7.7	
Grassland area change		1.3	17%
Agricultural land		6.4	

The price increase of maize generates a reduction in maize demand for animal feed of 18% of 115.4 ton = 21 ton, while also for other animal feed inputs the demand is reduced as a consequence of increases in crop prices, making in total a reduction of feed demand of 30. The production increase of 60.6 ton is accommodated for 28% by a yield increase, and for 72% by an increase in area[14].

Price elasticities

The total shock leads to a price increase of maize of 4% in the EU and 0.40% globally. Other price changes are not mentioned. With EU production being about 50 Mt, the increased maize production of about 10 MT (20%) implies a price elasticity of supply of about 20/4=5, while the reduction in demand of about 2.5 MT (5%) implies a price elasticity of demand of -1.25. Based on the yield effect the supply elasticity can be decomposed in a yield elasticity of 1.4 and an area elasticity of 3.6.

Reallocation effects

[14] However, it is mentioned that the additional feedstock production is not 8 MT, but at least 9.6 MT in Europe and 1 MT in Latin America, i.e. 75% instead of 56% of gross feedstock production of 14.2 MT. This may be explained as follows. DDGS is a substitute of a bunch of animal feed commodities of which maize is only one. Therefore, the increase of 3.7 MT DDGS production reduces maize production with only 1 MT, where the other part is substituted by other crops like soybean. If this interpretation is correct, it would have been beneficial if this would have been made explicit.

According to the calculation in Mt and including the yield increases 5.9 ha/TJ would be required. However, production requires 7.7 ha/TJ, implying that through reallocation land use requirements increase with 30%. This may be explained by lower yields of soy as one of the substitutes for DDGS. It is suggested that the 30 ton of extra DDGS supply per TJ substitutes with 5 ton of other protein meals, and 23 ton grains.

Land cover changes and GHG emissions

The increase in cropland results in increased pressure on land and therefore a reduction in grassland, where the reduction in grassland area is 17% of the increase in cropland area. The change in cropland is partly at the cost of abandoned land and natural land in the EU, but the reduction of soy meal production as a consequence of substitution with biofuels generates also a reduction in soy oil production that is compensated by an increase in palm oil production generating extra peatland oxidation.[15]

Table 16 Land use change and GHG emission per TJ maize ethanol. **Source:** Own elaboration.

	ha	gCO_2/year
Nat land	-5.7	9
Abandoned	-2.0	6
Peatland	-0.1	2
Cropland	7.7	-4
Grassland	-1.3	1
Total		14

Meat and milk production

The reduction in demand for animal feed and the reduction in grassland area suggests that meat production is reduced or that animal productivity increases. Nothing is told about this in the report.

[15] The analysis is a little bit more complicated, because it is mentioned that 110 kha of grassland in the US is given back to natural vegetation. It seems that this is because of a reduction in soybean area required, implying that the expansion of cropland into grassland in North America (as shown in the graph on land conversion) is in fact an expansion of cropland through grassland into natural area. Also in Latin America it suggested in the graph on land conversion that cropland expands into grassland, while the text suggests that no cropland expansion takes place.

Comparison with IFPRI study

Table 17 Analysis of Searchinger of the IFPRI maize ethanol. **Source:** Searchinger (2015)

	ha/TJ	
Gross feedstock area	13.5	
Co-product	6.0	44%
Net feedstock area	7.6	
Cons reduction	3.9	52%
Area without yield change	3.6	
Yield effect	2.7	75%
Crop area expansion	0.9	
Grassland reduction	0.4	43%
Agricultural land	0.5	

Table 18 shows the decomposition of the IFPRI maize ethanol case. Compared with the Ecofys study the gross feedstock area required is a little bit lower, while co-products seems to save more area, but if we include the reallocation effect it is more or less the same. Reduction in demand is much larger in the IFPRI study, and it seems mainly to be crop consumption reduction where in the Ecofys study it is only feed consumption. The yield effect is much larger in the IFPRI study as is the fraction of expansion into grassland in total cropland expansion. In the end, cropland expansion is much smaller in the IFPRI study.

Table 18 Calculation of land cover change and related carbon emissions based on Laborde (2011). **Source:** own elaboration

	Carbon stock		
	tCO_2/ha	area/TJ	tCO_2/year
Carbon from mineral soil (grassland)	100	0.40	2.0
Carbon from commercial forest	250	0.48	6.0
Carbon from nat forest	300	0.00	0.0
Peatland	1100	0.05	2.8
Total		0.88	10.8

The analysis of land cover changes (based on Table A8 of the report) shows again that although the total GHG emissions per TJ for maize are not fundamentally different in the two studies, the background of them is. The difference in GHG emissions per TJ is even smaller if we correct the numbers for the reduction in animal feed respectively consumption demand: IFPRI has 22.4 gCO_2/MJ, and Ecofys 20gCO_2/MJ, with for example cropland expansion respectively 1.88 ha/TJ and 10.9 ha/TJ, i.e. IFPRI having a cropland expansion that is 10% of Ecofys.

In summary, it is not easy to get a precise insight into the land use dynamics based on the information provided in the report. However, the rough dynamics is clear. Ethanol from maize generates a complicated substitution process because of DDGS as a by-product that replaces grain and soybean production and grassland for feed, where the reduced production of soybean oil explains the increase in palm oil production with its peatland conversion. Although the GHG emissions per MJ are consistent with those of the IFPRI (2011) study, the mechanism behind it is completely different.

Interpretation

Also Valin et al (2015) provide some messages on the interpretation of results of economic ILUC studies. An important message is that current emissions are high, but can be reduced if in the important regions a strict environmental policy is followed with respect to deforestation and peatland drainage (p. xv). However, such a policy is only effective if restrictions on use of natural forests or peatland are also applied to food, feed and other non-biofuel uses and also for supply to other regions than the EU, because otherwise the policy will have large leakage (p. xii).

ILUC can only be modelled, not measured, because it is generated by global mechanisms in a large interdependent system (p. V). The outcomes are the results of assumed causalities in the model (P. xiii), i.e. the understanding of the agricultural market system (p xv) and on biophysical values. For both it is very difficult to reduce uncertainty. As a consequence of this inherent uncertainty that cannot be avoided, it is not suitable to include ILUC factors directly in the calculations. Only large differences between groups of biofuels, like biodiesel, ethanol and second generation biofuels may be useful (p. xv).

For forest residues there is a lower build-up of soil organic carbon, and so it is not LUC emission in the strict sense of the word. Also for overharvesting of straw this effect may happen (p.xi).[16]

With respect to foregone sequestration the following comment is made: sequestration

[16] What happens with the straw that can be sustainably harvested if it is not harvested? Is it emitted to the air or groundwater?

may not happen because of annual mowing in order to receive CAP money, occasional mowing by smallholders or extensive grazing (p. xiii).

Crop prices will cause a decline in food consumption. This is not only because people eat less, but may also be because higher prices are an incentive for wood waste reduction (p. 2).

Comparison with the IFPRI (2011) study

For both the IFPRI(2011) and the Ecofys(2015) study co-products and yield increases are important (P xiv). Drainage for oil palm plantation expansion is also relevant for especially biodiesel effects, but also through DDGS co-products. The peatland effect is larger in the Ecofys-study, because estimated share of expansion into peatland is revised upwards and peatland emissions are revised based on the literature. This was already announced in the IFPRI study.

LUC in IFPRI is only 1.7 Mha, and 8.8 Mha in Valin for the EU biofuel mix, where also total emissions are higher, but emissions per crop are more or less the same, except for palm oil and soybean because of the revised peatland assumptions (p. xiv-v). Some further explanation are (p. 96):

- More palm oil and soybean oil is used than in Laborde
- No sugar can used in Valin study, but a lot in Laborde
- Emission factors are non-linear in GLOBIOM; increase with the size of the mandate. The weighted sum of single feedstock LUC equals 88g CO_2/MJ versus 97 g CO_2/MJ for the composed shock: two times more deforestation
- Table 12 gives an interesting comparison; large scale expansion generates: more cropland increase and short rotation plantation area is larger; grassland reduction less, less abandoned land decrease, also expansion in non-forest natural land is less and therefore more area expansion into forest.
- Much larger uncertainty ranges identified. With explanation.

In section 3.3.9 we have shown that the cause of the differences in results can be relatively precisely calculated.

Evaluation by the European Biodiesel Board

The **European Biodiesel Board** wrote a **reaction on the GLOBIOM study (EBB 2016)**. They complain that very little is explained about the model and the baseline, and that the model is not publicly available. EBB perceives that product-specific ILUC is higher than in the Laborde study, and that total ILUC is five times higher. Especially for soy and palm biodiesel the results are much higher than in other studies. The palm yield is low compared with the literature being 2.6 T/ha compared with 4 T/ha, where for rapeseed the oil ratio seems to be too high (54% instead of about 42%). For some reason the rapeseed market seems to be much more local than the sunflower market, where the difference in global prices versus EU prices seems in both cases surprisingly high: an increase of rapeseed production of 15% generates a price increase in the EU of 25%, while for sunflower an increase of production of 20% generates a price increase of 8%. Also the difference in global prices for different types of vegetable oil seems to be high, with especially small effects for palm oil prices.

The most surprising effect in the GLOBIOM study is that an increase in biodiesel from sunflower and rapeseed generates an increase in the consumption of meat and milk as a consequence of lower feeding cost, while in other studies it reduces meat and milk consumption. According to the EBB this is the consequence of an unrealistic strong reaction of protein prices in animal feed.

The biodiesel board also recognizes that low yield elasticities in simulations are inconsistent with a baseline where yield increases are high compared with production increases. This is however consistent if you assume that most yield increases in the baseline are exogenous and not depending on prices.

With respect to the sensitivity analyses the EBB perceives the sensitivity analysis as unclear, where especially for yield elasticities they are surprised that the range compared with the average is mentioned, but not the size of the elasticities.

Another comment is on the concept of "foregone sequestration" of abandoned land in the EU that is a large part of GHG emissions of EU biofuels.

A provocative opinion piece by Kotrba (2016) of the EBB provides some further comments on the Ecofys approach. One is that it is surprising that ILUC of candy and sweet goods companies is not analysed, while consumption of them is bad for health. A reply on this could be that this production is not subsidized while biofuel production is. Even though there may be arguments to increase taxes on candy and sweets, politically there is a fundamental difference between not taxing externalities and subsidizing commodities.

An interesting observation in the same direction is that if farmland in the US would be converted to forest land, this would also have ILUC effects that may destroy the benefits of this conversion on a global scale. A third comment is that a lot of other factors than biofuels determine LUC, like for example in Indonesia the use of forests for the wood and paper industry that may stimulate illegal logging and land clearance.

A fourth comment is on accountability: why would the biofuel producer be responsible for effects they can't manage? If Malaysia would implement appropriate safeguard systems, LUC for biodiesel would be much smaller, according to the GLOBIOM study.

Conclusion

Despite all efforts to give decompositions in the report by Valin et al (2015), it is difficult to trace down the precise stories behind the different biofuels.

4.3.4. ICCT's guide for the perplexed (2014)

Approaches

Before starting with the general analysis based on economic approaches, **ICCT (Malins et al. 2014)** discusses the non-economic approaches. A historical approach is Fritsche et al. (2010) that is based on historical patterns of trade flows and cultivation expansion, and assumes the same pattern for all land expansion, implying that ILUC differences depend only on differences in yield (p. 37). However, the example of palm oil suggests that relations are more on a global scale.

A causal-descriptive approach is Bauen et al. (2010) (p. 37) where a few trading and land use patterns are assumed to be dominant, and historical and expert knowledge is used to trace them. The disadvantage is that more complex issues are not tackled, and that different experts may provide different results. They refer also to historical statistical analysis that is however not mature enough to get some conclusions (p. 37).

Determinants of ILUC emissions

The main part of the ICCT-study is the **decomposition of ILUC demand** in its main components, being the price elasticity of food demand, the price elasticity of yield, crop switching, location and trade, the utilization of co-products, the price elasticity of area expansion, the question which land is converted into cropland, and the carbon stocks of converted and agricultural land. These elements are discussed in this section.

Price elasticity of food demand.

Roberts and Schlenker (2010) suggest based on econometric historical analysis that demand response is about equal to the supply response to price, implying that between a third and a half of feedstock for biofuels is accomplished through consumption reduction (p. 45). ERS-USDA price elasticities suggest that poorer regions reduce agricultural consumption more when prices increase, so the effect of price increases will be larger for them (p. 53), and therefore the welfare impacts of food consumption reduction may differ.

Price elasticity of yield

Relevant for yields are baseline yield, the marginal yield and the yield elasticity (p. 56). History shows a linear growth of maize yield (and therefore a decrease in percentage yield growth over time), and a lot of variation in yield per metric ton per ha in the world (p. 57). Countries with low yields have structural problems that are not easily overcome, explaining why the yield gap is not closed over time in practice (p. 58). Therefore the yield gap will not automatically be closed when biofuels demand is introduced. For example, some farmers may prefer low-yielding crops because they don't have fertilizer enough, or because they need more straw for their animals. There is a hint that the linear yield growth may slow down because the most easy yield improvements have been realised and perhaps because of climate change (p. 58). The lower the trend rate in yield, the higher ILUC may be in the future. Yield growth is defined as the growth of the average yield and therefore includes both yield change as a consequence of price increases and yield change because yield on new land differs from average yield (p. 59).

Some yield changes are long-lasting and others are reversible. For example, increased fertilizer or labour use may be reversed when prices go down, while effects of extra R&D are more long lasting (p. 60). Increased fertilizer use increases N_2O emissions may according to Edwards et al. (2010) cancel out GHG benefits[17], while more intensive farming may also generate erosion and decreased carbon sequestration of soil.

High prices may stimulate R&D and investment in higher yields, and for example EC DG Energy (2010) argue that because of this increased biofuel demand may pay itself back through increased productivity (p. 60-1). This may be consistent with the cyclic agricultural investment that (HLPE 2011) finds, but (Liu & Shumway 2007) find no correlation, and even argue that low prices may stimulate yield by forcing farmers to adopt new technologies at the risk of bankruptcy. Therefore, the conclusion is that the

[17] But according to Laborde (2011) these effects are small.

hypothesis that biofuel demand will induce a green revolution is “anything but clearly confirmed from the existing literature.” (p. 61)

Public funded research is historically the main driver for innovation, and it may be argued that research is much more effective in boosting yields (p. 62). In this context it is mentioned that US subsidies on biofuels are six times the annual R&D budget of Monsanto.

Berry(2011: p. 8) also has serious doubts on the large yield effects that are included in some economic biofuel models. He argues that price elasticity of supply comes almost only from land-use, not yield where also farm-level studies don’t find yield effects (p. 14). Roberts and Schlenker (2010: 8) don’t find a correlation between yield shocks in a given year what you would expect if it was caused by prices. Also long-term historical evidence (p. 57, figure 3) doesn’t show a correlation between price and yield changes, although this may also be caused by the fact that in the past government policy had a more important role compared with market forces in guiding farmer decisions (p. 66). Berry and Schlenker (2011) do an econometric analysis with instrumental variables and show that the price elasticities of yield are small and not significantly different from zero (p. 66). Anyhow, they are much lower than the 0.25 used by GTAP and 0.2 used by MIRAGE, whose values are based on according to Berry inadequate econometric evidence, not based on instrumental variable methods, as discussed in Keeney and Hertel (2008: p. 20) and a working paper by (Huang & Khanna 2012, p. 15).

Long term yield elasticities may be larger than short term elasticities, but the same may be true for area elasticities (p. 64), and because the relative size of the elasticities matters, it is not clear what this implies. Yield elasticities may differ by region (p. 65).

Double cropping is sometimes used as an argument for yield increases as a consequence of higher prices. However, in Brazil it seems to be more like an autonomous process, while in US there is small anecdotal evidence based on one crop and one price shock (p. 67).

The net yield effect is a combination of changes in average yields on current area and the yield on new area, the marginal yields. Analysis based on set-aside land gives different estimates, while estimates using a land productivity model for maize gives about 10% lower productivity on new fields. However, it may be that infrastructure and other considerations determine where crops are grown, creating even the possibility of higher yields on new land. Most models use marginal yields that are 10 till 50 percent lower than average yields (p. 68-72).

Crop switching, location and trade

Location of expansion of crops, both biofuel feedstock and other crops, is important because differences in location imply differences in yield and differences in carbon stocks of new land, where the effect of changes in location is in the same order of magnitude as yield increases (p. 79). In economic models one assumption may be that the cheapest location is used, given transportation cost and tariffs. By the way, the cheapest location is not automatically the location with the highest yield. Another is the Armington assumption, i.e. the share of demand satisfied by imports is fixed except when relative prices change. This implies that demand increases tend to be met with a fixed share domestically and trade patterns will not easily change (p. 75-6).

Armington elasticities can be calibrated using historical data, but in reality they are mostly not. If elasticity data are published, it is difficult to test the quality of them; they may be based on different periods or different regions, (p. 36)[18].

Next to substitution between different production locations, also switching between different crops is important. For example, substitution between rapeseed oil and palm oil is high (p. 78).

Utilization of co-products

Co-products may have an important LUC reducing effect. Table 3.2 (p. 80) gives an overview of co-products for different feedstock, and makes us aware that also electricity (for cellulosic biofuel and sugarcane ethanol) may be relevant for the GHG balance, even though it is not included in ILUC. Most of these effects by the way may be included in LCA analysis.

The analysis of the effect of co-products is complicated. First, one has to determine the co-product yield per unit of biofuel. Second, one has to determine which feed components are displaced, and finally one has to determine what the land use

[18] And we may add, may be estimated in a general way, while they may depend on a lot of circumstances like the share in trade for Armington elasticities

requirements of the displaced feed components are. For example, if DDGS replaces soy instead of wheat, this requires more area and has more GHG consequences (p. 81). The amount of co-product produced is relatively uncontroversial, but what is exactly substituted and its consequences for land use and carbon are uncertain. Complex models suggest that substitution is complicated (p. 82). For example in Laborde (2011) direct substitution between soy and DDGS in the model results not in a reduction in soy, but protein feed is substituted with other feed. However, at least in GTAP modelling it is shown by running the model both with and without co-products that co-products in the end have significant effects on land use (Edwards et al. 2010).

What feed is replaced is determined either by feed trials, i.e. experiments, by comparing nutrient content or by a least cost programming approach (p. 82). (Hoffman & Baker 2011) give an overview of field trial studies, which according to CARB(2010, p. 31) provides estimates that are too high because the starting point is a suboptimal diet (p. 83). The comparison of nutrients method has also problems because of necessary simplifications. For example Lywood, Pinkney and Cockerill (2009) only investigate protein and energy content, where only wheat and soy are used as substitutes (p. 83).

The third approach is least cost programming, where feeding cost is minimized given current prices (Klasing 2012; Hazzledine et al. 2011) and the stylized conclusion is that 1 ton of DDGS replaces roughly 1 ton of other feed (p. 84). In some models, (including GLOBIOM and CAPRI), least cost programming of feed ratios is included explicitly in the model making the replacement ratios consistent with prices in the scenarios.

ICCT conclude that the land requirements for displaced feed require deeper consideration, and that one must be aware that the substitution possibilities depend on the type of livestock. The idea that DDGS is simply replaced by a combination of soy meal and wheat/maize is too simple. Because soy meal requires more area, the replacement of DDGS by soy may have important consequences for ILUC (p. 86).

For biodiesel co-products the conclusions are more or less the same. Oil seed rape meal has a lower palatability than soy meal and phosphorus content is higher, and has a higher fibre content making it difficult to digest for non-ruminants (p. 86). Laborde (2011) shows that substitution can be affected by market conditions next to nutritional values: there is a substitution towards feeding animals more protein when the price of protein meals fall (p 88, table 3.8).[19]

Price elasticity of area expansion

First, we must acknowledge that area expansion depends on a combination of yield and area elasticities, where area elasticities are suggested to be much larger than yield elasticities according to econometric analysis with instrumental variables by Roberts and Schlenker (2010). However, a problem is that area switching towards other crops is not distinguished from general area expansion in Berry and Schlenker (2010) and Roberts and Schlenker (2011). Regionally comprehensive estimates of area expansion do not exist, to the best of the knowledge of Malins et al (p. 92).

Another issue, discussed already with respect to yield changes, is that crop acreages used to depend to a large extent on government policy (p. 90) and that only recently market faces became more important (Taheripour et al. 2011, p. 11). Laborde (2011a) uses elasticities taken from developed countries based on historical data (Laborde and Valin, 2012; Golub, 2006) and further assumes that they are higher in developing countries where policies to control deforestation and land conversion are weaker and less well enforced (p. 91). Using the same area elasticities around the world would generates the wrong areas of expansion in the world (p. 92).

Which land is converted into cropland

When it has been determined how large the expansion of cropland or total agricultural land is, the basic questions is what type of ecosystem is destroyed. The first step is to determine the fraction of cropland expansion into each type of land, the so-called land extension coefficients (LEC) (p. 97). These LEC’s may be determined by comparing satellite data on land cover, as is done by Winrock-MODIS (p. 97). Their approach is criticized because there is much uncertainty in satellite data. When 5% of area is

[19] It is obvious that this is the result of the highest CES nest in the feed production structure.

incorrectly allocated and when this is random, almost 10% of land use changes that are measured may be wrong; this is a multiple of actual real land use changes (p. 97). Miettinen et al (2012) give more robust results with much more precise satellite data. However, MIRAGE uses Winrock-Modis LECs to allocate land expansion over land categories (p. 98).

A second approach to allocate land expansion is accomplished in models like MIRAGE and GTAP where a CET function determines based on relative prices to what extent cropland expands into commercial forest and commercial grassland (p. 97). This approach is never used to analyse expansion into pristine land, because for this land it is only possible to allocate prices in an artificial manner.

Model results show (p. 98-9, figure 3.11) that they tend to allocate more land expansion into grassland than into forest. Forests store more carbon than shrub land and shrub land more than grassland. GTAP used for CARB has only grassland and managed forest to expand in, MIRAGE also has no shrub land, while FASOM uses 25 different forest species types and 18 forest management intensities, so this may influence the carbon consequences.

A third approach to the problem of the allocation of cropland expansion may be the use of a land allocation model, where land characteristics per grid cell like rainfall, slope, soil quality, proximity to roads and distance to existing production areas determine the probability of land conversion (p. 98). Malins et al refer in this context to work of the Joint Research Center (Hiederer & Ramos 2010), but for other studies also for example the land allocation model CLUE or IMAGE have been used.

Carbon stocks

If it is known which types of land are being converted it has to be determined what the consequences for GHG emissions are. Models like MIRAGE and GTAP tend to use average carbon stocks per unit of each type of land cover, but if there is some unknown systematic bias for agriculture to expand in certain types of land, this may have large consequences for carbon stocks involved (p. 100). It is important not only to consider carbon in biomass, but also carbon in soil, that tends to be more in grassland than in forest. Furthermore, also deadwood and litter in forests are important, because this contains about 10% of carbon stock in tropical and 40% in temperate forests (Litton et al, 2004; Turner et al, 1995; Delaney et al, 1998).

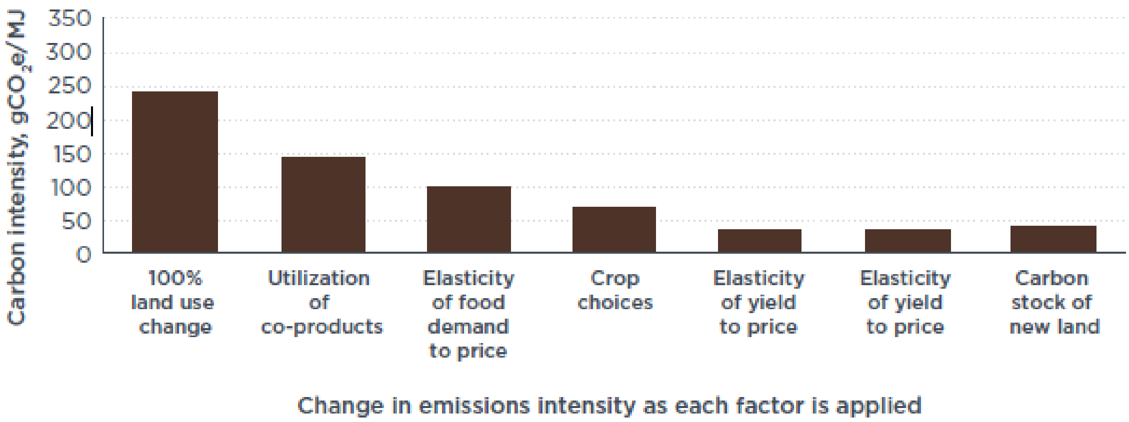
Even when one knows how much carbon stock there is in the ecosystems, one has to find out how much is emitted in the atmosphere. (Searle & Malins 2011) find that 10% in developed world, and 3% in developing world of cleared biomass remains stored in harvested wood products and landfills after 30 years (p. 101). Also important is the question to what extent the transformed land would have sequestered carbon over time. This foregone sequestration is especially relevant for abandoned land (p. 102). Furthermore, also carbon stocks in cropland is relevant, as some biofuel crops (soil plus biomass) may store more carbon than the original land converted (p. 102).

Peat soils may require specific attention (p.103), because oxidation as a consequence of peatland drainage is an important and long term source of greenhouse gas emissions. Peatland emissions are much higher than originally estimated. Laborde (2011) show how important assumptions about peatland emissions are for the GHG results, and we may add to this that Valin et al (2015) confirms these results by implementing the higher estimates in their model.

Simple decomposition analysis

GTAP-BIO US-corn ethanol decomposition

The analysis is applied to the results for US corn ethanol of the GTAP-BIO model (p. 112) and for an uncertainty analysis based on a simple model derived from the analysis above (p. 107-110). The results of the GTAP-BIO model are roughly consistent with the analyses that have been shown for the MIRAGE model above as their figure 3.4 that is reproduced below shows.



Note: Because GTAP is usually referenced with regard to the California Low Carbon Fuel Standard, which uses a 30-year emissions amortization, a 30-year amortization has been used here. JRC (Edwards, Mulligan, and Marelli 2010) presented 20-year results.

Figure 30 Effect of various factors on the emissions reported by GTAP for U.S. corn ethanol. **Source:** Malins et al (2014)

The Witzke decomposition

ICCT decomposes GTAP results using an analysis based on the (Witzke et al. 2010) decomposition. The basic idea is that the change in emissions depends on change in land use because of different yields after reallocation of land because of net trade, change in demand (biofuels and other demand) and changes in yield, weighted by land.

ICCT extends this analysis by decomposing land use change in the land required to meet biofuel demand minus the reduction in land requirement because of by-products, plus the change in land use because of consumption (negative normally) minus the reduction of land use because of change in yields, plus the land required or saved because of international reallocation of production.

Not So Reduced Form ILUC Model

To investigate uncertainty further, ICCT develops a simple ILUC model that is inspired by the simple land use model by Plevin et al (2010). This model has a very simple structure. It consists of the fuel yield, the net displacement factor, and emission factors of forest, grassland and wetland converted to cropland, with their fractions, and a production period to translate the one time conversion greenhouse gasses towards a per year base. Monte Carlo simulation is used with distributions based on highest and lowest estimates in the literature and insights in the type of distribution to reach a general distribution of ILUC factors. It is inspired by the large uncertainties, especially with respect to the economic models that predict the displacement factor.

A simple model is developed, also basically being a decomposition with some parameters included, where the basic idea is that everything is first calculated on a global level, and that then the regional deviations are included to adjust parameters. So, there is a parameter EL that represents the conversion of the average piece of land in the world to agriculture.

After an initial calculation on average values for land use and emissions and without corrections, land savings because of co-products, systemic efficiency improvements, difference between average and marginal yields, adjustment for average carbon stocks to the actual land used for expansion, and an explicit correction for peat emissions is added. See for the results section 3.4.4.

4.3.5. JRC study using the historical approach (2015)

The decomposition method into the historic analysis by Overmars et al (2015) is more or less consistent with the reduced form ILUC model excluding the consumption reduction part. First gross land use per TJ is calculated. Then they estimate the part of the area that is because of by-products, so they don't take into account for which feed exactly is substituted. The main step is that they assume that the distribution between yields and area is the same as the historical one, resulting on an area increase . They already mention themselves that they systematically underestimate the area effect. Searchinger et al (2015) argue however that the main part of historical yield increases is not price-based, but based on exogenous factors. Overmars et al mention that the area increases are sometimes negative, and in that case they take the absolute value of the area increase in the past in those cases. So, the historical analysis seems not to be a satisfactory foundation, because it is not the historical relationship between price and area versus yield change, but just the distribution of historical production change that is distributed over historical area and yield change, independent of the question to what extent production or price change are the cause of the yield change.

Two methods for the location of crops diverted by biofuels are used. First, the local approach, where it is allocated in the region where the biofuel is produced. As an alternative it is allocated to the regions that export the crop. Finally, emission factors applied to the land use changes are based on two possible land allocation models. These are emission factors per country (and maybe crop specific), i.e. it is not made explicit which types of land conversion are behind them. Next to the factors above, Overmars et al also investigate to what extent international allocation of feedstock influences the results. They just take the average yield differences, and multiply the share with these yield factors.

With respect to consumption effects in LUC analysis, these are not taken into account in the Overmars et al study. This may be seen as an advantage, because in this manner food security effects are not calculated as ILUC benefits for biofuels. It implies also that in comparing results of LUC with other studies the results without consumption have to be taken.

In summary, the historical allocation approach seems not useful as an approach to analyse ILUC, but parts of the simplified steps may be a useful source of inspiration in the context of simplified ILUC analysis, although not all steps are as lucid as you would like them.

4.3.6. GTAP-based analysis[20]

Analysis of EU biofuel policies (2013)

(Darlington et al. 2013) analysed EU **biofuel policies** with the **GTAP model** as an assignment of the European Biodiesel Board. One conclusion is that compared with the IFPRI(2011) analysis in the GTAP model less land would be converted and of converted land less would be converted from forests. The first reason behind this result is that marginal land in the EU has a higher productivity in GTAP than in MIRAGE (p. 19). Second, GTAP has a separate nest for pasture-crops versus forest with a lower elasticity, explaining the smaller share of expansion of cropland into forests. Another issue not mentioned as a reason in the report, is that they increased the price-yield elasticity from the default value of 0.25 towards 1 in order to take into consideration that cropland may be underutilized: "This is one simple way to model the utilization of some of the cropland that is currently fallow or in cropland pasture" (p. 24).

A second conclusion is that GHG emissions per ha are lower in GTAP. GTAP has an extra land cover category, cropland-pasture that is pasture land that has been recently used as cropland and therefore has less carbon stock difference with cropland (they assume half). The study also experiments with the idea that fallow land may be used and they just model it as if it were an increase in yields (because fallow land is still seen as cropland in statistics). And they also experiment with an assumption that no forest is converted in regions where no forest has been converted in the last decade (US, EU and Canada).

For the use of fallow land they show with EU-statistics that the increase in rapeseed area is accompanied with a reduction in fallow land, making according to themselves a strong case for their conclusion. However, be aware that foregone sequestration that is important in the Ecofys study is not included in their analysis.

With respect to yields the study observes that in the EU rapeseed productivity has not

20 CARB.Lowcarbonfuelstandardre-adoptionindirectlandusechange(ILUC) analysis, September29,2014.Availableat: ⟨http://www.arb.ca.gov/fuels/lcfs/lcfs_meetings/092914ILUC-prestn-color.pdf⟩

increased between 2004 and 2012 (despite wide differences in yields between countries) and that in Canada yields have been increased with 20% in the same period.

Another interesting observation is that both in the EU and the US there is a reduction in beef consumption (for EU production) per person, while poultry consumption is reduced during the last 20 to 30 years, and because of this substitution where beef requires 18 kg feed and poultry 3 kg per kg boneless meat the amount of animal feed needed per person has been reduced (p. 32).

One final conclusion is that more focus should be on effective restrictions on forest conversion, and on carbon losses or gains from putting cropland pasture and fallow land back into production.

The end result of the exercise is that this GTAP analysis even has smaller land use and greenhouse gas effects than the IFPRI (2011) analysis (see their table 9, reproduced below). The ILUC-factors are even around 60% lower in the GTAP-study for the EU compared with the IFPRI (2011) study.

Table 19 Land Use changes for Biodiesel Feedstocks. **Source:** IFPRI (2011)

Feedstock	This Analysis 8.6% (ha/1000L)	IFPRI2011 Values (ha/1000L)	Percent Change from IFPRI2011 to This Analysis
Soy Biodiesel	0.1036	0.1378	-27%
Palm Biodiesel	0.0499	0.0682	-33%
Rapeseed Biodiesel	0.1138	0.1392	-34%
Other Biodiesel	0.0526	0.1749	-70%

Table 20 Land Use Emissions of the Biofuels Scenarios. **Source:** IFPRI (2011)

Biofuel	IFPRI2011 Analysis, 8.6% Target (g CO_2/MJ)	% Change from IFPRI2011 to This Analysis
Palm Biodiesel	54	-56%
Rapeseed Biodiesel	54	-65%
Soybean Biodiesel	56	-50%
Other Biodiesel	52	-79%
Wheat Ethanol	Maize:10, Wheat:14	-33%*
Sugar Beet Ethanol	Beets: 7, Sugarcane:13	+136%*

* Estimated from the average IFPRI2011 emissions

If we compare this analysis with the Ecofys (2015) study, we see that these last issues have been included in the Ecofys modelling, but that this is compensated by a much larger increase of cropland conversion than in the IFPRI-study by having smaller relative yield elasticities and other factors discussed elsewhere in this report.

Gohin analysis of the cause of update differences

(Gohin 2016) analyses the causes of the reduction of GHG effects of soy biodiesel between two model revisions and the difference in results for canola biodiesel in the EU and the US. One important background is the change of GTAP database from version 2001 to version 2004. In general the yield-price elasticity has been revised downwards, and the yield-area elasticity (what is that?) upwards. The emission factors have been revised towards AEZ-specific EF's, with on average lower EF's in the US and Canada and higher in the EU and Brazil (p. 405).

However, the main reason for the difference between the two years is that animal feed trade increased in the database with as a consequence more substitution possibilities for animal feed and therefore smaller local price effects, while in the old database some countries like China don't produce oil meal as a co-product in the old database (p. 406). By correcting both issues in the old database, the results approach the results with the new model and database. The over-supply of animal feed in the US is reduced, implying less reduction in grassland and other animal feed inputs, while in other regions the supply of animal feed is increased implying that less crops are needed.[21]

For canola (rapeseed) biodiesel, the EU data in the GTAP database show differences in land rent per hectare, and as a consequence the CET function substitutes a different number of hectares. Especially, the land rent for "other agricultural products" is almost nine times as high as that for maize. This generates a large reduction of production of "other agricultural products" forcing a land use change also in South Africa. This problem is solved by equalizing the rents per hectare for different regions. A next problem is in Armington elasticities that are much lower for the EU than the US because internal trade competes also in the Armington function and this problem can be solved by deleting internal trade. A third problem is that oil meals from canola seem to be too low in the EU (17% instead of 29%). The combination of these

[21] Although I don't understand that the size of this effect is so big, where crop area expansion outside the US is five times crop area expansion in the US, and this is reduced to 50% of this in the new version.

revisions reduces the ILUC effect of the EU towards that of the US (with about 26 g CO_2e per MJ).

Decomposition of LUC

Hertel and Baldos (2016, p. 103) based on Hertel et al (2010) show a decomposition of LUC of corn ethanol in the US. They start with the corn ethanol conversion factor and corn yield generating the amount of land needed. Then reduction of use of corn for exports and animal feed and the use of DDGS in animal feed reduces required corn area with 60%. Then increased yields reduce required area with 17%, but this is compensated by a lower yield on new corn area of 12%, implying a small yield effect on area.

Then Hertel and Baldos (p. 107) generalize the decomposition towards all land requirements by doing selectively less restrictive simulations. The reduction in non-food demand and intensification of livestock and forestry reduces land requirements with 25%. By-products reduce it further with 30% of the original amount of land use. Reduced food consumption adds another 15 percentage points of reduction, where yield increases on current land compensate more or less the lower yields on new land, implying that in the end about 25% of land area required for ethanol is converted from forest and pasture.

However, in newer runs of GTAP the yield elasticity has been increased, implying that the LUC effects in GHG emissions seem to be better. Searchinger et al (2015) calculate a decomposition of LUC for US wheat ethanol according to GTAP in CO_2 equivalents per MJ, which generates with the high yield elasticities a reduction of 40 g CO_2 per MJ, and as a consequence of consumption reduction an extra 32 g CO_2. According to this calculation even in this high yield elasticity case maize ethanol has only a GHG benefit compared with fossil fuels because of this consumption reduction effect, while the balance would be negative in case the consumption effect would not be allocated to the biofuels (Searchinger et al, 2015, p. 1421).

4.3.7. GoVilla project (2016)[22]

Approach

The project **GoVilla** (Governance zur Verminderung von indirekten Landnutzingsänderungen) provides **options for governance to reduce indirect land use effects** (Schebek, 2016a, p. 16-7). The focus is on local government options to reduce **ILUC in Brazil, Indonesia and Ukraine**, including the international context, and including the analysis of political structures and actors (p. 35). The scenarios developed are a combination of international governance, regional governance (Indonesia, Brazil, Ukraine) and **EU/German biofuel policy scenarios** (p. 29).

One part of the research describes available biofuel technologies in Germany and policies in Brazil, Costa Rica, Indonesia and Ukraine. These form the background of scenario analysis that is accomplished with a combination of the global general equilibrium model MIRAGE and the land use model LandSHIFT (p. 30). MIRAGE has been improved by including multi-cropping, the inclusion of new multi-product technologies/sectors, improvement of classification of land categories and substitution possibilities of land, and possibilities to use fallow land (p. 30-1). LandShift is used to allocate regional production to grid cells by creating a multi-criteria value function and allocating land use based on this (p. 32-3). Based on this grid cell allocation greenhouse gas emissions can be calculated per grid cell (p. 34), but also changes in average land productivity can be calculated that are fed in as exogenous technological changes in MIRAGE, while MIRAGE feeds LandShift with endogenous yield changes generated by substitution processes between land, fertilizer and non-land production factors.[23]

Baseline and scenarios

All scenarios are focused on 2020.[24] The baseline shows a yearly increase of yields of about 2% per year, with sugar cane and palm fruit around 3% per year. Agricultural

[22] http://www.govila.tu-darmstadt.de/govila_govila/publikationen_8/publikationen.de.jsp
[23] Technical Paper I-2.
[24] The next part is based on Technical paper I-4 of the CoViLa project

prices show a slight increase. The baseline shows an increase in cropland area of 40 mln ha in 10 years, and a reduction in grassland of 4 mln ha. In Brazil cropland expands at the cost of grassland, where in Sub Saharan Africa pasture area increases. Primary forest area is reduced with 93 mln ha, where forest plantations increase with 15 mln ha. So, forest area is reduced more than the expansion of agricultural area.

Scenario S1 analyses the effect of an EU biofuel demand of 5.14 Mtoe bioethanol and 16.1 Mtoe biodiesel. As a consequence agricultural production increases with 1.87%, where livestock production is reduced with 0.1% and food prices increase with 0.25% (p. 8). Regretfully in the reporting nothing is formulated per TJ of biofuel, the land needed for area is not presented, and the total shock is not presented. So, we don't go more in-depth.

Scenario S2 analyses the effect if ILUC coefficients calculated in IFPRI(2012) are used, resulting in 5.17 Mtoe bioethanol and 0.14 biodiesel Mtoe demand by the EU (p. 15). This scenario implies that biofuels are mainly produced from sugar cane and beet, where sugar cane growth negatively impacts maize and soybean production, mainly in Brazil, by attracting land and investments, and in Europe the additional sugar beet production for ethanol generates by-products used to feed livestock, and helps to replace the maize and soybean meals (p. 16). Emissions are in this case reduced to 2.9 CO_2eq/MJ, significantly lower than in S1, but also than in Laborde (2011/2).

Scenario S4 is a scenario where biofuels in the EU are completely phased out.

Environmental governance as mitigation option

These biofuel scenarios are combined with governance scenarios. The point of reference is the Business As Usual scenario (A). The Best International Climate Governance (B) implements strong land use policies globally and increases in productivity in Brazil, Indonesia and Ukraine and a somewhat higher biofuel consumption than in the baseline. The other scenarios are Business as Usual with only for Brazil respectively Indonesia Best Climate Governance, and a Medium International Governance scenario (E) (Chapter 4 and p. 160).

With respect to Brazil there are opportunities to increase productivity of pasture land, and therefore if improved government prevents expansion into forests and other natural vegetation (especially in the Cerrado) greenhouse gas emissions can be reduced a lot.

Indonesia has a lot of potential to reduce emissions by a moratorium on concessions for primary forests and peatland, the implementation of national and regional action plans to reduce greenhouse gas emissions combined with increased efficiency in administration. Maybe increased use of abandoned and degraded land is possible.

With respect to land use in Ukraine there is a lot of abandoned land that has already accumulated a lot of carbon.

If all measures are implemented, ILUC can be reduced only a little bit (table 5.29, p. 181), by a stricter governance, i.e. the most strict governance scenario B has only 2 g CO_2/MJ less emissions than in the Business As Usual scenario.

We have to be aware that the land use changes in MIRAGE are relatively small compared with other models because of the high yield elasticities compared with area elasticities.

4.4. Are there trends in feedstock-specific ILUC factors?

Results of recent ILUC studies are far from consistent in their outcomes, and after 2012 there seems to be no further convergence in results. In table 22 below we must be aware that the CARB estimates are for the US, implying that they could be different. However, (Gohin 2014b) suggested that when GTAP calculates the effect of European rapeseed oil the differences in their results were the consequence of problems in the database and that after correction the US and EU results were almost the same. Therefore, it seems that the table below gives an indication of the differences in studies.

Table 21 ILUC comparison per feedstock (gCO_2-eq/MJ) between 5 studies. **Source:** own compilation

	Laborde (2011, p. 78)	Valin(2015)	CARB(2009)	CARB(2015)	GTAP-EU (2013)
wheat	14	34			10
maize	10	14	45	30	7
sugar beet	7	15	0	0	16
sugar cane	13	17	69	18	32
rapeseed oil	54	65	63	22	19
soybean oil	56	150	95	44	28
palm oil	54	231	0	107	24
* US outcomes; Adjusted towards a 20 year period					

The basic conclusion must be that especially with respect to biodiesel the change from results around 2010 and results in 2015 have a different direction in the US compared with the EU: while Valin (2015) has significantly higher results for soybean and palm oil than Laborde (2011), CARB reduced its estimates for biodiesel emissions. Furthermore, we have seen that even with comparable results between the Laborde (2011) and Valin (2015) study the mechanisms behind these results are fundamentally different with Laborde having mainly land use change in regions far away, and Valin mainly forest reversion emissions and other natural land conversions in the EU.

In interpreting the results above, one should be aware that the results include the effect of reduced consumption, and therefore if one would like to exclude this from the ILUC factors, one should increase the ILUC factors with 30% to 50%.

4.5. Conclusion

The in-depth analysis of some well-known and policy-relevant ILUC studies shows that within the group of economic models the essential difference is not between partial and general equilibrium, but about the question what mechanisms are included in the model. However, although the mechanisms are not fundamentally different, the outcomes are and the main mechanisms behind the end results are. For example, if you compare the land use changes of the recent CARB estimates for corn ethanol with those of Valin (2015) and Laborde (2011), Valin has a cropland use change of about 0.9 ha/TJ, Valin of 7.7 ha/TJ and CARB(2015) of about 5 ha/TJ. Despite the difference in land use expansion, the GHG emissions don't differ much between the Valin and Laborde study, while CARB has more or less double the land use emissions from those calculated by Valin despite that the area effect is smaller. In summary, even emission factors that look more or less the same have a completely different dynamics.

Analysing results from different reports is very labour intensive, and in the end it is impossible to derive the main mechanisms from the reports. This is something recognized also by for example Searchinger et al (2015), Tyner et al (2016) and Malins(2014). It is essential to open the black boxes behind the reporting of the model results.

The essence of the results can be explained by a limited number of shares and other parameters, that are related with combinations of model parameters. For example, the consumption effect depends on demand, yield and area elasticities, where the percentage of production increase accommodated by yields depends on the area and yield elasticities together.

This implies that interpretation of the literature should be based on the main explanatory parameters that are distilled out, more or less like in the ICCT study, and the analysis performed in sections 3.2.-3.4 of this report. We have seen in chapter 4 that this is not an easy task requiring further investigation.

5. Conclusion

Since 2012 there is **not a systematic evidence of convergence of ILUC factors** found in the literature. Especially with respect to the ILUC effect of biodiesel estimates by CARB have a different direction compared with those in the EU. The much higher values found in the EU (Valin, 2015) are mainly based on the assumptions that increases in vegetable oil production will through substitution processes in the end mainly be through increases in palm oil production, where also based on recent investigations it is suggested that peatland oxidation generates more carbon emissions, while also a larger percentage of new oil palm plantations will be on peatland.

When **decomposing results**, one would like to have information on the land use changes as a consequence of feedstock production, co-products, reduced consumption of both crops and livestock, yield increases in feedstock, other crops, livestock and commercial forestry, and the emission factors, split into the distribution of land use expansion in natural areas and the carbon emissions related with them. In presenting the ILUC factors, one would like to have also a value without the consumption effects, because these should be attributed to the reduction in consumption instead of to the biofuels.

The analysis of the empirical evidence on the different components of ILUC shows that for most steps in the **decomposition empirical evidence is extremely poor**. Even evidence on long term supply and demand elasticities of agricultural commodities is not available, because almost all elasticities are estimated based on annual changes and are therefore short term in character. In analysing supply and demand one should be aware that the reactions may be very complex, including for example policy reactions on high prices. In most models the combination of demand and supply elasticities reduces ILUC with 30% to 50%. However, if you follow Searchinger et al (2015) the ILUC reduction because of reduced consumption should not be allocated to the ILUC factor of biofuels, and in that case the consumption effect is not that crucial any more.

The share accommodated through **yields** is determined by a combination of the yield and area elasticity. Also for these elasticities, being the components of the supply elasticity, the information is very meagre. Recent econometric evidence suggests yield elasticities to be zero or only a little bit higher, but again, these estimates are all short term and in most cases crop specific. Long term yield elasticities may be higher than short term yield elasticities because investments to increase yields require time, but they may also be lower when in the short term unsustainable options for yield increase are used that cannot be continued in the long term. Factor substitution, fertilizer use, and especially the role of autonomous technological development are very difficult to disentangle in the empirical analysis of yield increases.

With respect to **area** elasticities, these are mostly estimated together with the yield elasticities and are normally at least as high as the yield elasticities. For area elasticities it is also plausible that the long term elasticities are higher than the short term elasticities, because area expansion requires investment. Furthermore, area elasticities tend to be smaller in regions where land is scarce.

The **relocation effect** is about substitution between different agricultural commodities and different areas. Especially international relocation of **production** implies in many cases different yields, where the international relocation depends on the modelling of trade. Two main approaches to model trade are the Armington approach and the international world market approach The Armington approach assumes that future trade depends on current trade that may be increased or reduced by relative price differences. The integrated world market approach assumes that price effects are the same for all regions, implying that production expansion takes place in those regions where production can most easily be expanded. The Armington approach tends to allocate production more in the neighbourhood of the region where demand is generated, and for the US and the EU this implies that the effect on expansion of agricultural land outside these regions is less. On the other hand, if adjustment costs in trade are sufficiently high in the integrated world market approach, as is for example the case in GLOBIOM, then the reaction maybe more like in the Armington approach except for that not current trade patterns are the crucial factor determining trade.

Also the allocation of **cropland expansion** on different types of land can have fundamental consequences. Empirical evidence about these patterns is also very meagre, implying that different models can have totally different allocations. Even if deforestation happens, the question is to what extent it is caused by cropland expansion or other factors. Brazilian evidence suggests that illegal logging in combination with illegal expansion of livestock because of old pasture area becoming degraded, are important drivers of deforestation in Brazil. Improvements of legal enforcement have reduced deforestation a lot in Brazil.

Finally, the **emissions per type of land conversion are uncertain**. Different accounting systems exists, and discussion is about correct categorisations of land conversion types. For example, GTAP uses a land use category cropland-pasture that is pasture land that used to be cropland and therefore has a lower carbon stock than normal pasture land, while Valin et al (2015) assume that if abandoned land is not converted into cropland it will sequester carbon over time.

Based on the overview of the lack of knowledge of the ILUC components we may conclude that a **systematic analysis of uncertainty** is extremely difficult. The standard method of analysis is a Monte Carlo analysis, but there is no empirical foundation of the ranges and distributions used. In most cases, the Monte Carlo analysis gives ILUC ranges that are from negative to much too high to generate GHG savings, and therefore are not very informative, while many authors recognize that also the outliers of the Monte Carlo analysis could be relevant. Therefore, for example Tyner and Taheripour (2016) decided not to do a Monte Carlo analysis for the investigation of uncertainties, but to investigate the effect of a limited number of parameters separately.

In investigating the results of different models, it is difficult to track the precise background of the results. This is consistent with the conclusion of Persson (2016, p. 479) that "far too many studies, simply focus on the quantitative outputs (e.g., price changes) of single model runs without attempting to understand or explain the model dynamics that give rise to those results, compare outputs with empirical data, or conduct parametric and structural sensitivity analyses."

Precise analysis such as accomplished by Gohin (2016) of recent GTAP results for the EU shows that imperfections in the database or in implicitly chosen parameter values may have disastrous effects on the outcomes. The in-depth study that we performed on the study by Valin (2015) shows that even behind the same ILUC factors completely different mechanisms can be hided. It is not easy to trace the mechanisms easily from published papers and reports.

Section 3.5 investigated **strategies to reduce ILUC**. The first strategy focuses on **low ILUC feedstocks**. One type of low ILUC biofuels is to produce them from co-products like straw and forestry residues. There seem to be opportunities from an ILUC perspective, but one has to take into consideration that:

- Harvesting residues may be at the cost of organic soil carbon
- Harvesting residues may provide incentives to switch to techniques with lower productivity for the main products
- Harvesting residues for biofuels may be at the cost of using them for other

purposes.

- Harvesting residues for biofuels must also be cost effective

The second strategy is to **grow feedstock on marginal lands**, i.e. land that is not used for other purposes. When perennials are used on degraded or low carbon land that would not be used otherwise, the carbon value of the biofuel feedstock may be higher than the carbon value or carbon sequestration potential in the original vegetation, generating negative emissions from land use change. However, be also aware that the marginal land could also have been used for the production of other commodities like paper pulp that may reduce production of these commodities elsewhere reducing the pressure on pristine areas or releasing agricultural land.

A third strategy is to **increase yields**. Several studies suggest that investment in R&D and extension services has high returns. However, if you require these investments to certify biofuel production, it is basically a conditional sale. So, if these policies are useful for biofuel production, why wouldn’t you apply them also to food production?

A fourth strategy is the **protection of areas with high carbon stocks**. An important aspect is that policies to avoid conversion of natural vegetation are not necessarily the result of the use of biofuels or policies that stimulate the use of biofuels. In other words, the benefits of protection of natural vegetation and lower ILUC emissions from food and biofuels production cannot be allocated to the production of biofuels only, unless these policies are implemented as part of the policies that stimulate the sustainable production and use of biofuels. Moreover, the protection of natural vegetation may limit the ILUC emissions of biofuels, but this may also lead to a trade-off with higher food prices and higher impact on food consumption.

In general it can be concluded that the **certification** of low ILUC and ILUC free biofuels is unlikely to be able to avoid all indirect effects. Additional measures, beyond the scope of certification, are therefore needed, such as integrated land use planning including territorial policies.

The approach described in section 3.2 shows a method to decompose results that however requires that models provide region-specific insights into the production and land use effects of each biofuel pathway, including the substitutions assumed in animal feeding. An interesting approach with respect to by-products could be to model a change in supply of by-products as a separate experiment next to one with both the biofuel production increase and the increase in by-products, or to model the biofuels increase with and without co-products.

The decomposition approach discussed in sections 3.2 and 3.3 is the way forward in analysing model results of ILUC and on gathering information on empirical knowledge that is relevant for ILUC. Current reports give at the most a very impressionistic picture of the forces that reduce indirect land use change, and for none of the reports it was possible to derive a decomposition as suggested here. Therefore, for future reporting the calculation methods should be better defined and harmonized, as suggested in deliverable 1, and the method developed in this report that is inspired by (Searchinger et al. 2015) and IPCC(2014) seems to be a useful approach for this.

However, making the background of modelling results explicit is something else than improving the empirical knowledge that is incorporated in these results. It is key to get better insights into yield elasticities, the location of area expansion, substitution processes in animal feeding generated by increased biofuel co-product supply, and the types of land that are going to be converted as a consequence of cropland or agricultural area expansion.

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Appendix 1: Summary of Land Based Carbon

1. Introduction

In order to get a better understanding of the latest information on land-based carbon stocks we investigated the parameterization of the most influential studies, the literature on which they are based, and other relevant reports/literature. Most of the studies focus on forest carbon stocks as these are among the most important sources of LUC carbon emissions. Comparison of the parameters across literature is hampered by different definitions of land types and regional aggregations. Below is a summary of the carbon stocks for different land types as identified by key studies. Particular attention was paid to the uncertainty of these carbon stocks as well as the ranges of c-stock distributions.

2. Forests

From the studies investigated, the most detailed c-stock information is provided for forests. Comparison across studies in order to develop a *single*, or even a *single set*, of forest carbon stocks is not straightforward since different studies have different definitions and aggregations for forest lands. These include managed, unmanaged, accessibility, conversion likelihood , etc. Furthermore, regional aggregation also inconsistent, which may cover countries, regions, sub-regions or agro-ecological zones. Below we summarize the identified values of carbon stocks of living and dead biomass as well as soil carbon.

2.1. Living and Dead Biomass Carbon

Guayana

The Guyana Forestry Commission published a report outlining the potential greenhouse gas emissions resulting from deforestation and forest degradation (Goslee et al. 2014). Their results are based on field data collection using the sampling design plan and other factors from the Intergovernmental Panel for Climate Change (Aalde et al. 2006). The purpose of the study is to investigate emissions factors for deforestation driven by mining, agriculture and infrastructure. The land strata used are aggregated as follows:

- High Potential for Change, More Accessible (HPfC-MA)
- High Potential for Change, Less Accessible (HPfC LA)

- Medium Potential for Change (MPfC)[25]

The study also incorporates uncertainty in biomass carbon poo ls and soil by investigating the propagation of error as well as a Monte Carlo analysis (see section 5 below). The results are summarized in **Error! Reference source not found.**.

Table 1 Parameter Carbon stocks by pool for the three strata in Guyana (tC/ha). Minimum and maximum based on 95% confidence interval as determine from error propagation (% of mean, HPfC-MA: 7.8%, HPfC-LA: 10.1%, MPfC: 12.1%). See Section 5. **Source:** Goslee et al. 2014.

	Above Ground (tC/ha)	Below Ground (tC/ha)	Saplings (tC/ha)	Dead wood (tC/ha)	Litter (tC/ha)	Total (Min-Max) (tC/ha)
HPfC-MA	193.6	45.5	4.2	13.1	3.3	259.8 (249.7-269.9)
HPfC-LA	267.6	62.9	4.1	10.8	5.6	351 (333.3-368.7)
MPfC	231.1	54.3	3.5	7.9	3.2	300 (281.9-318.2)

Laborde 2011

Appendix II of the study of Laborde (2011) provides carbon stock values used by the MIRAGE model. These are disaggregated across 11 world regions and 18 Agro-Environmental Zones (AEZ). **Error! Reference source not found.** shows C-stocks for managed and primary forests across 11 world regions. The average, minimum and maximum are determined by comparing the different AEZs in each region. It is important to note that while carbon stocks vary across AEZs, they do not vary across regions for a given AEZ. Thus differences in regional numbers are due to the presence of different AEZs. These numbers have not been weighted according to the share of each AEZ per region. Original data is available in Laborde (2011), Tables A3 and A4.

Though the publication claims the values are in tCO_2/ha, the numbers would make

[25] Carbon stocks for "More Accessible" and "Less Accessible" MPfC lands are not significantly different (P=0.9) and thus these two strata were combined into a single MPfC.

more sense if they were in tC/ha. Unfortunately, the report does not cite a primary source and thus it is difficult to verify this. It is also not clear what carbon pools are included in these numbers.

Table 2 Carbon stocks in managed and primary forests (tCO_2/ha). Based on AEZ per region (out of a possible 18 AEZs). **Source:** Laborde (2011).

	Managed Forest C-Stock (tCO_2/ha)			Primary Forest C-Stock (tCO_2/ha)		
Region	Minimum	Maximum	Average	Minimum	Maximum	Average
Brazil	134	354	232	291	708	400
Central America & Caribbean	0	0	0	291	708	460
China	34	354	173	34	708	275
CIS	34	294	149	34	463	214
EU27	34	294	172	269	269	269
Indo-Malaysia	134	354	247	291	708	459
Latin America	34	354	149	112	708	329
Rest OECD	34	354	175	34	269	159
Rest of the World	34	354	153	112	463	253
Sub-Saharan Africa	68	354	200	159	708	334
USA	34	294	149	34	463	214

Valin et al. (2015) / FAO (2010)

Valin et al. (2015) uses estimates from the Global Forest Resources Assessment (FAO 2010) which is largely based on country reports of forest resources. The methodology used to determine c-stocks is based on the IPCC, however it is noted that for some parameters (such as carbon fraction) different countries may use older guidelines (i.e. IPCC 2003 instead of 2006). The numbers in Table 42 of Valin et al. (2015) are different than those reported in Table 2.21 of the FAO (FAO 2010). It is unclear why the difference arises. **Error! Reference source not found.** reproduces Table 2.21 of the Global Forest Resources Assessment.

Table 3 Carbon stock (tC/ha) in forests by region and subregion, 2010. **Source:** Valin et al. (2015)

Region/subregion	Carbon in Biomass (tC/ha)	Carbon in Dead Wood and Litter (tC/ha)	Total Carbon Stock (tC/ha)[26]
Eastern and Southern Africa	58.9	14.6	119.4
Northern Africa	22.2	8.8	66.0
Western and Central Africa	116.9	10.2	186.2
Total Africa	82.8	11.7	145.7
East Asia	34.4	7.2	109.4
South and Southeast Asia	85.6	3.6	145.1
Western and Central Asia	39.8	12.6	89.0
Total Asia	60.2	5.8	125.7
Europe excl. Russian Federation	63.9	18.6	179.1
Total Europe	44.8	20.5	161.8
Caribbean	74.4	14.8	149.2
Central America	90.4	36.6	185.4
North America	55.0	38.5	151.8
Total North and Central America	56.1	38.2	152.7
Total Oceania	54.8	15.3	113.3
Total South America	118.2	11.6	217.1
World	71.6	17.8	161.8

2.2. Soil Carbon

Guyana

The report of the Guyana Forestry Commission also estimated soil carbon stocks as well as the impact of land use change on them. Stocks after conversion were estimated based on land use, management, and input factors as derived from the IPCC (Aalde et al. 2006). By combining the data in **Error! Reference source not**

[26] Includes carbon in soil, shown in **Error! Reference source not found.**.

found. and **Error! Reference source not found.**, the Guyana forestry commission concludes that the LUC emission factor from agricultural expansion in forests is 57.1 tCO_2/ha for HPfC-MA, 72 tCO_2/ha for (HPfC-LA) and 64.2 tCO_2/ha for MPfC.

Table 4 Carbon stock, Carbon Stock change and soil emissions for the three strata in Guyana (tC/ha) for soil Carbon. Minimum and Maximum values calculated based on 95% confidence interval as a % of the mean (HPfC-MA: 21.6%, HPfC-LA: 17.4%, MPfC: 21%), see Section 5. **Source:** Goslee et al. 2014.

	C-Stock (tC/ha)	C-Stock after 20 years of agriculture (tC/ha)	Soil Emission (tC/ha)
HPfC-MA	99.3 (86.3-112.3)	47.7 (41.4-53.9)	51.7 (44.9-58.4)
HPfC-LA	80.3 (73.3-87.3)	38.5 (35.2-41.9)	41.8 (38.1-45.4)
MPfC	96.5 (86.4-106.6)	46.3 (41.5-51.2)	51.2 (44.9-55.4)

Laborde 2011

As with carbon stocks, Appendix II of Laborde (2011) provides carbon emissions from mineral soil (tCO_2/ha) in Table A 5. Again these are displayed for 11 global regions and 18 AEZs. Unlike carbon stocks, soil emissions vary both per region (due to differences in AEZ presence per region), but also within a given AEZ across regions. These are displayed in **Error! Reference source not found.**, where the average, minimum and maximum are determined by comparing the different AEZs per region. As with living and dead biomass, there is reason to believe the unit of these numbers is tC/ha, instead of the reported tCO_2/ha).

Table 5 Carbon emissions from mineral soil (tCO_2/ha). Based on AEZ per region (out of a possible 18 AEZs). **Source:** Laborde (2011)

Region	Minimum	Maximum	Average
Brazil	56	113	79
Central America & Caribbean	57	112	89
China	27	103	69
CIS	36	108	73
EU27	37	108	77
Indo-Malaysia	46	93	65
Latin America	28	107	70
Rest OECD	9	108	48
Rest of the World	26	104	63

Sub-Saharan Africa	28	113	73
USA	34	108	73

Valin et al. (2015) / Repo et al. (2015)

Valin et al. (2015) uses estimates of soil from sustainable harvesting of logging residues, *not* land use change! These are based on Repo et al. (2015), and the carbon losses estimates are dependent on the decomposition time of soil litter which is a function of temperature and precipitation. These emissions are estimated for 25 European countries and they range from 0-6 tCO_2/ha over 20 years, and 1-10 tCO_2/ha over 50 years.

FAO (2010)

The Global Forest Resources Assessment (FAO 2010) provides numbers for carbon in soil, reproduced in **Error! Reference source not found.**.

Table 6 Carbon stock (tC/ha) in forest soil by region and subregion, 2010. **Source:** (FAO 2010).

Region/subregion	Carbon in Soil (tC/ha)
Eastern and Southern Africa	46.0
Northern Africa	35.0
Western and Central Africa	59.1
Total Africa	51.1
East Asia	67.8
South and Southeast Asia	55.9
Western and Central Asia	36.6
Total Asia	59.6
Europe excl. Russian Federation	96.6
Total Europe	96.4
Caribbean	60.0
Central America	58.4
North America	58.4
Total North and Central America	58.4
Total Oceania	43.2
Total South America	87.3
World	72.3

3. Peatlands

The use of peatlands has been identified as a major source of emissions. Laborde

(2011) states that the emission factors in literature range from 50 to 120 tCO_2/ha, having increased among more recent studies. Laborde uses an EF of 55tCO_2eq/ha.yr (increasing from 19 tCO_2/ha.yr of the previous MIRAGE report).

Valin et al. (2015) conduct a detailed review of peatland emissions. They estimate peatland emissions factors based on the following determinants:

- The level of water table (drainage depth)
- Natural respiration variability and timing of the measurement
- Peat bulk density and the fraction of carbon in the soil
- Measurements used to estimate fluxes in GHG. Namely measurements of soil subsidence, direct flux measurement through closed chambers, and measurements by Eddy covariance

The studies considered were all peer reviewed. Furthermore, in the case of closed chamber studies, they only considered those separating autotrophic (root respiration) and heterotrophic (peat oxidation) calculations as this provide a bias that can play a significant role around trees.

The emissions depend on different uncertain multiplicative drivers, among which oxidation rate, peat bulk density, and subsidence rate, which is related to the water table. As there is no large scale dataset on the distribution of these factors over the regions of the study, the authors assume a log-normal distribution. This profile is confirmed by observation with flux chambers. Thus Valin et al. (2015) reproduce a distribution profile consistent with the literature. The mean distribution is 61 ± 22 tCO_2/ha.yr, the median is 58 tCO_2/ha.yr and the confidence interval at 95% in the range 27-112 tCO_2/ha.yr. The first quartile of the distribution is at 44 tCO_2/ha.yr, which is in the magnitude of the Tier 1 value of IPCC (2013). The third quartile is 74 tCO_2/ha.yr, above most closed chamber measurements but below some measurements on acacia plantations which tend to have higher emission factors.

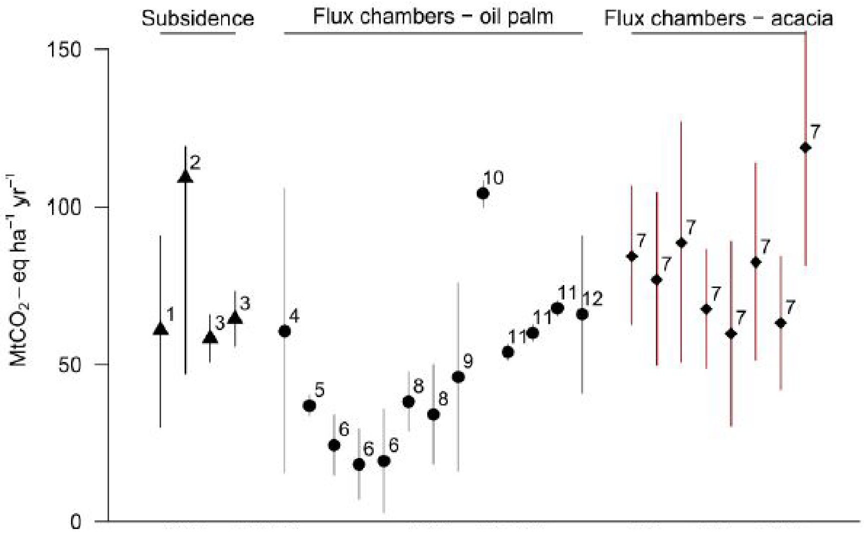

Figure 1 Distribution of central estimates of studies and range of uncertainty. **Source:** Valin et al. 2015.

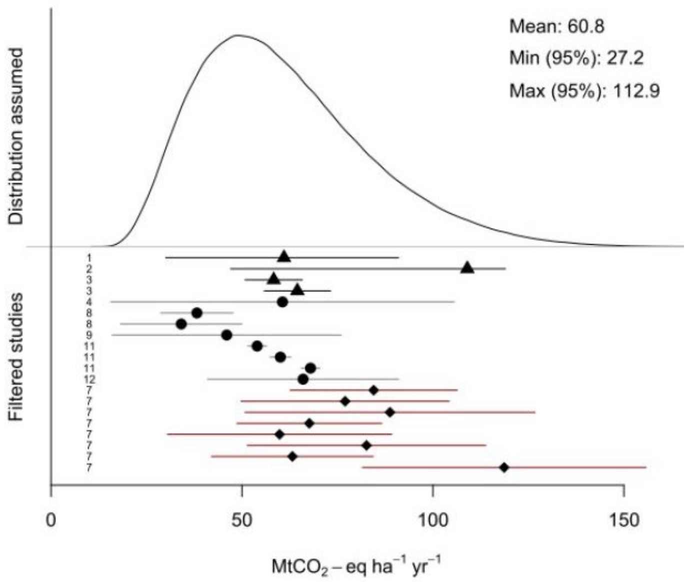

Figure 2 Peatland emission distribution and ranges provided in literature. **Source:** Valin et al. 2015.

4. Other Lands

While most studies focus on forest areas, Valin et al. (2015) provides carbon stocks for other natural lands and grasslands as well. These are based on Ruesch and Gibbs (2008), who produced a global map of biomass carbon stored in above and belowground living vegetation using IPCC methods (Aalde et al. 2006). The carbon stocks for Grasslands and Other Natural Lands used in the GLOBIOM model are summarized in **Error! Reference source not found.**.

Table 7 Average carbon stock values used in GLOBIOM. **Source:** Based on Ruesch and Gibbs *(Ruesch & Gibbs 2008)*.

Region	Above and Below Ground Biomass (tC/ha)	
	Other Natural Land	Grassland
Latin America	26	7
South Asia	29	3
North America	10	3
EU28	9	3
Eastern Asia	14	2
Southeast Asia	29	5
FSU	4	4
Sub-Saharan Africa	35	4
Oceania	17	3
Middle East and North Africa	13	2

5. Major Sources of Uncertainty

The effect of uncertainty is only specifically addressed by the report of the Guyana forestry commission as well as the IPCC guidelines. The Guyana report estimates uncertainty in biomass carbon pools (ranging 7.8% to 12.1%) and soil carbon (17.4% to 21.5%), with uncertainty varying across land strata. Uncertainty in fire emissions was not estimated given its insignificant contribution (<0.2% of total emissions). The report determines the overall uncertainty in carbon stocks using two methods: Error propagation and Monte Carlo analysis. The results are summarised in **Error! Reference source not found.**.

Table 8 Comparison of uncertainty estimates for biomass stocks (living, dead and soil) using error propagation and Monte Carlo Analysis. **Source:** Guyana Report

Statum	Uncertainty, 95% confidence interval as a % of mean	
	Error Propagation	Monte Carlo
HPfC-MA	7.8%	8.4%
HPfC-LA	10.1%	14.3%
MPfC	12.1%	NA

The report states that "Monte Carlo may be appropriate because correlation will exist between various measured carbon pools and between estimates of carbon stocks developed at different points in time. Using Monte Carlo rather than error propagation method improves estimates of uncertainty from a Tier 2 to a Tier 3 method.

Though Laborde (2011) does not specifically investigate uncertainty in carbon stocks (while uncertainty in other parameters is studied), two major issues are highlighted:

1. What is the right average value of carbon stocks per hectare in a region? Does the use of averages (as done in this report [and elsewhere]) induce a bias?
2. Among all source (sic) of emissions, the case of palm trees grown on peatland is amongst the most sensitive for our results.

Valin et al. (2015) included peatland emissions (described in Section 3 above) in their uncertainty analysis. Furthermore, Repo et al. (2015) (Used by Valin for soil carbon loss estimates) highlights that the largest sources of uncertainty in the simulated changes of soil carbon stocks are litter input estimates. This is particularly true for litter production of fine roots and branches which is poorly known. Nevertheless, it is claimed that the general conclusion are not sensitive to these uncertainties.

The IPCC Guidelines for National Greenhouse Gas Inventories (Aalde et al. 2006) highlights that uncertainty for forest carbon factors include basic wood density (10-40%), annual increment, land-use management and reference soil c-stocks. Ultimately uncertainty arises in methods that determine carbon stocks using Tier 1 which ignores annual changes in carbon stocks. However, the resulting error could be small on a landscape level as increases in some stands could be off-set by decreases in others.

No uncertainty assessment is presented in the Global Forest Resources Assessment (FAO 2010).

6. Conclusions

The numbers presented above indicate that carbon stocks vary widely across studies. Total forest carbon stocks (biomass and soil carbon) range from 360-431 tC/ha in Guyana (Goslee et al. 2014) and 66-217 tC/ha regionally (FAO 2010). Studies set their uncertainty to approximately 20% (95% confidence interval), with this being higher for peatlands.

Interestingly, while most studies broadly agree with each other, the numbers used my Laborde (2011), are very different, unless the unit is quoted incorrectly. Laborde presents all numbers as tCO_2/ha, while they would be in broad agreement with other studies if they were tC/ha.

As expected, most studies trace their methods back to IPCC good practices, with variations in results of the studies arising due to different geographic aggregations (socio-economic, country, regional, agro ecological-zone) and the availability of newer (local) primary data.

Most studies, including the Valin (2015) study that is relatively explicit, are not very clear in the fundamental mechanisms that explain GHG emissions because of LUC. A clear decomposition of results, as suggested in section 3.2. and applied as for as possible in chapter 4, is an essential requirement for an open discussion of LUC effects of biofuels.

However, the analysis of the empirical knowledge of the different components of land use change shows that fundamentally too little is known. Therefore, with respect to research the focus should be more on generating further knowledge on the different components of land use change than on simulating with complicated models that hide the limited information on which they are based.

Appendix 2: Summary of Biomass Carbon [27]

1. Agriculture Biomass

A potentially important land-based carbon flux is that associated with "agricultural biomass". These are fluxes associated to the carbon content in agricultural living biomass (cropland) when agricultural activities expand. This flux can be negative (sequestration) or positive (emission). The former may take place if palm trees expand into grasslands, and the latter if sugarcane is replaced by lower c-content soybeans.

Valin et al. (2015) accounts for this flux for 11 feedstocks according to the LUC projections of the GLOBIOM model. Carbon stock variation is subsequently divided by 20 years in order to obtain annual emission flows. According to Valin, this flux is the main negative emission source concerning biomass production, and their results are presented in Table 9. According to the GLOBIOM LUC projections, for all crops there is an increase in carbon stocks due to biomass growth, with the largest gain in Palm oil. However, this does not lead to an overall lower Palm Oil emission factor since it has the highest LUC emissions. Overall, C stocks in biomass leads to a reduction in the overall emission factor by approximately 20% for 1st generation biofuels, but plays a very important role for 2nd generation biofuels, with c-stocks in agricultural biomass being responsible for the negative overall emission factors projected by Valin et al. (2015). Table 1. Annual emissions from agricultural biomass and foregone sequestration for 11 biofuel crops. Data A-D taken from Figures 3&5 and section 4.3 to 4.25 in Valin et al. (2015). All energy units in secondary (biofuel) terms

[27] In order to make things comparable: one ton of carbon equals 44/12 = 11/3 = 3.67 tons of carbon dioxide.

2. Foregone Sequestration

	Data from Valin et al (2015)					*Own Calculations*			
Feedstock	Agricultural biomass emission factor (gCO_2/MJ)	B. Foregone sequestration emission (gCO_2/MJ)	C. Total LUC (gCO_2/MJ)	D. Energy Productivity 2020 (GJ/Ha)		Emission reduction due to C in biomass: ((C-A)-C)/(C-A)	Emission fraction of foregone sequestration: B/C[28]	Agricultural biomass sequestration (tCO_2/Ha)	Foregone sequestration (tCO_2/Ha)
Wheat Ethanol	-5	12	34	42		13%	35%	0.21	0.50
Maize Ethanol	-4	6	14	64		22%	43%	0.26	0.38
Barley	-6	11	38	38		14%	29%	0.23	0.42
Sugarbeet	-2	4	15	145		12%	27%	0.29	0.58
Sugarcane	-15	0	17	118		47%	0%	1.77	0.00
Sunflower oil	-12	7	63	24.5		16%	11%	0.29	0.17
Palm oil	-90	0	231	88		28%	0%	7.92	0.00
Rape oil	-12	15	65	52		16%	23%	0.62	0.78
Soybean oil	-25	1	17	150		14%	1%	0.43	0.02
Perennial	-12	8	-12	90		100%	40%	1.08	0.72
Short Rotation Plantations	-30	5	-29	97		3000%	15%	2.91	0.49

[28] For "Perennial" and "Short Rotation Planation's", since the overall emission factor is negative, the fraction of foregone sequestration is calculated via B/(B-C).

It is also important to account for C-stock increase which may have taken place in theIt is also important to account for C-stock increase which may have taken place in the absence of biomass production. For instance, biofuel production may lead to reduced afforestation or reduced return of cropland to natural vegetation. This effect takes place in particular in Europe where a trend exists of cropland abandonment (Valin et al. 2015). For the GLOBIOM projections, this emission source is also shown in Table 1, where it has been annualized over 20 years. Foregone sequestration accounts for 0-43% of the emission factor of 1st generation biofuels and is most important for maize ethanol.

Daioglou et al. (2016) use the IMAGE-LPJmL model in order to determine the changes in land based carbon stocks over time. Thus, they determine foregone sequestration for 6 different land types, for an RCP 2.6 climate projection, and annualized the c-stock changes over 20 years. The results are shown in **Error! Reference source not found.**. As expected grasslands have the lowest foregone sequestration, since they have overall low carbon stocks, while future growth in forests leads to significant sequestration. Abandoned agricultural lands lead to the highest foregone sequestration since they would otherwise revert to natural lands, with much higher C-stocks than agricultural lands.

Table 9 Foregone sequestration emission factor (annualized over 20 years) according to the IMAGE-LPJmL model. **Source:** Daioglou et al. (2016)

Land type	Foregone sequestration emission (tCO_2/Ha)
Grasslands	0.59
Boreal forests	1.15
Tropical forests	4.03
Temperate forests	2.65
Savannah	1.88
Abandoned Agricultural lands	4.61

It is important to note that the IMAGE-LPJmL analysis does not include land-use projections and merely investigated the c-stock changes in different land types. Thus, though the results indicate higher foregone sequestration emissions in forests, GLOBIOM projections do not project such land use changes. IMAGE-LPJmL results concerning Grasslands (and to a lesser extent, Savannahs) are in line with GLOBIOM results. Furthermore, while LPJmL is in line with other similar models, there are indications that these models overestimate this effect, partly due to not taking into account limits on Nitrogen.