A causal descriptive approach to modelling the GHG emissions associated with the indirect land use impacts of biofuels

Final report

A study for the UK Department for Transport

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E4tech

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List of acronyms

1G	First generation
2G	Second generation
CAGR	Compound annual growth rate
CARD	Center for Agricultural and Rural Development
CHP	Combined Heat and Power
DfT	Department for Transport
EAG	Expert Advisory Group
EC	European Commission
EU	European Union
FAME	Fatty Acid Methyl Ester
FAO	Food and Agriculture Organisation of the United Nations
FAPRI	Food and Agriculture Policy Research Institute
FASOM	Forest and Agricultural Sector Optimisation Model
GHG	Greenhouse gas
GTAP	Global Trade Analysis Project
HFCS	High Fructose Corn Syrup
HVO	Hydrotreated Vegetable Oil
IEA	International Energy Agency
ILUC	Indirect land use change
IBGE	Instituto Brasileiro de Geografia e Estatistica
ICONE	Instituto de Estudos do Comércio e Negociações Internacionais
LCFS	Low Carbon Fuel Standard
Mtoe	Mega tonne of oil equivalent
OECD	Organisation for Economic Co-operation and Development
PSD	Production, Supply and Distribution
R&D	Research and Development
RED	Renewable Energy Directive
RFA	Renewable Fuels Agency
RFS 2	Renewable Fuel Standard 2010
RTFO	Renewable Transport Fuel Obligation
SADC	Southern African Development Community
UNEP	United Nations Environment Programme

UNICA	União da Indústria de Cana-de-açúcar
UN-REDD	United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries
USDA FAS	United States Department of Agriculture – Foreign Agricultural Service
US EPA	United States Environmental Protection Agency
WEO	World Energy Outlook

1 Executive summary

This report summarises the outcome of a study commissioned by the UK Department for Transport (DfT). The study aims to develop an understanding of the chain of causes and effects that lead from an increased demand for biofuel feedstock to indirect land use change (ILUC), and provides a framework for capturing and quantifying those relationships. It specifically studies the greenhouse gas (GHG) impacts associated with the land use changes identified. Although there are clearly other important environmental and social impacts associated with land use change, these impacts were outside the scope of this study. The study is based on a causal-descriptive methodology which uses cause and effect logic to describe and derive the ILUC impacts, and makes wide use of stakeholder input. It provides an alternative modelling approach to the equilibrium models that have been increasingly used for ILUC factor calculations for biofuels, and could potentially be used to inform those models.

The causal-descriptive methodology used required mapping of all the impacts a biofuel has on the broader agricultural and land use systems, in order to identify all the possible land use changes that a biofuel can cause. The indirect land-related impacts of the biofuel (in terms of additional GHG emissions) were then established by estimating the quantity and type of land use change which occurs as a result of different market responses resulting from biofuel demand. The cause and effect relationships were estimated using a combination of extrapolation of historical trends, input and validation on future markets by an expert advisory group, and stakeholder feedback. This has provided a more transparent and participative approach compared to the equilibrium modelling approaches used to date, whose underlying data and assumptions are more difficult to access and thus limit the potential for stakeholder participation.

The methodology used does not explicitly model prices and as such differs from equilibrium models. Price modelling is inherently uncertain and would have added significant additional complexity beyond the scope of this project. However, prices have been implicitly considered by using supply projections based on historical trends and expert opinion to understand deviation from historic trends, as this was considered the best proxy for what might happen in the future.

This study estimates the ILUC impacts of five different biofuel feedstocks: palm oil, rapeseed oil, soybean oil, wheat and sugarcane. The scope was limited to these five feedstocks as they were considered by DfT to be the most relevant to the UK, in view of the fact that these are the main feedstocks used for biofuels consumed in the UK. US corn ethanol was therefore not covered in this study.

For each feedstock, several different ILUC factors are calculated. These scenarios represent different assumptions regarding the context and cause and effect relationships. Different scenarios are required because ILUC modelling is complex and uncertain for several reasons. Firstly, the modelling requires projecting impacts in the future, which is inherently uncertain. Secondly, the ILUC models cannot be validated or calibrated against historic data because indirect land use change is not an observable parameter, meaning that several potential impact pathways may be possible. Finally, there are uncertainties associated with the carbon stocks of different land types and the carbon stock losses associated with land use change. A “central” ILUC factor is not provided because of this large degree of uncertainty. A central ILUC factor may also detract from the message that ILUC impacts in 2020 will very much depend on the decisions that are made now about how we mitigate against ILUC.

This study finds that the size of the ILUC impact of biofuels can indeed be large, but varies significantly depending on the feedstock used and the future context considered. For example, if palm oil plantations continue to expand onto high carbon stock land (forest land or peat land), the risks of ILUC are large. Depending on the scenario assumptions made in this study, the ILUC factor calculated ranges from 5.9 to 82 g CO_2e / MJ palm biodiesel. In order for the impacts to be at the lower end of the range, effective policy to protect high carbon stock land and prevent expansion of palm onto peat land is required.

The ILUC impacts associated with rapeseed biodiesel are lower than for palm biodiesel in this analysis, but still potentially significant. In the scenarios examined, the ILUC impacts range from 15 to 35 g CO_2e / MJ. Scenarios in which more rapeseed is imported from outside the EU result in higher ILUC factors. This is due to (a) a difference in yield (OSR yields in the EU are higher than yields in the Ukraine and Canada, leading to larger amounts of land used for OSR cultivation for a similar amount of biodiesel in the latter countries), and (b) the type of land use change caused by oilseed rape production (on average the land in Canada has slightly higher carbon stocks than in the EU).

The ILUC impacts associated with soybean biodiesel in this analysis are somewhere between the ILUC impacts of palm oil biodiesel and rapeseed biodiesel. This is because, according to our analysis, an increase in demand for soy oil will not lead to an increased area of soybean grown, but to substitution of soy oil in other (e.g. food) markets by other oils, such as rapeseed oil and palm oil. The rationale for this is that because the soy oil is such a small proportion of the bean, it is not the main determinant of its value and therefore production, i.e. soy oil does not drive soybean expansion. Depending on the importing market, we have assumed a different mix of substituting vegetable oils. The soy biodiesel ILUC factor is therefore a product of the palm and rapeseed ILUC factors and varies in the scenarios explored in this analysis from 8.7 to 66 g CO_2e / MJ. One implication of this is that reducing the ILUC impacts of soy relies on reducing the LUC impact of other vegetable oils, palm in particular. Therefore, it is more difficult to identify practices in soy cultivation that lower the ILUC factor associated with soy oil biodiesel.

The modelled ILUC impacts of wheat ethanol are much lower than the other biofuels. In the scenarios looked at, they range from -53 to -5.1 g CO_2e / MJ. This is mainly due to the large credit given to wheat bioethanol by assuming that wheat DDGS is used as an animal feed. The negative ILUC factors in this case indicate that the ILUC factor would result in an ILUC credit for wheat rather than a debit. The scenarios with a larger ILUC credit are associated with scenarios in which no change in the current EU trade balance for wheat is assumed. However, the more realistic scenarios are perhaps those in which the EU wheat trade balance changes (e.g. a decrease in European exports or an increase in European imports). The scenario in which the EU sees a decrease in exports is associated with a smaller ILUC credit, but the total ILUC factor is still negative. This is an interesting outcome and one which highlights the ILUC benefits of using biofuel co-products to replace other land based products (e.g. animal feed).

Sugarcane ethanol ILUC impacts are modelled here to be in the range of 7.8 to 27 g CO_2e / MJ. Despite the fact there are no co-product "ILUC credits" attributed to the sugarcane ethanol chain, the ILUC impacts associated with expansion, displacing crops or pasture land onto "new" land, are low compared with other feedstocks considered, as a consequence of the typically high ethanol yield per hectare. The magnitude of the ILUC impact will depend on future trends in deforestation, pasture displacement and pasture intensification. Given the high impact that national policies can have on these effects, we explored several possible scenarios. Lower ILUC impacts are estimated in scenarios in which a lower deforestation rate is assumed. As for palm oil biodiesel, where expansion occurs onto forest land, the emissions from land clearing are attributed to the biofuel. This may be considered to be a conservative approach, as certainly at least a part of the reason for deforesting will be for the timber from the logging activity. Lower impacts are also observed if higher than historical pasture intensification rates are observed in regions where pasture is being displaced to. Pasture intensification is an area for further research, as its drivers are complex and the GHG implications still unclear.

It was not possible within the scope of this project to adequately model the additional indirect emissions associated with the increased use of fertiliser associated with yield increases. However, some preliminary calculations show that although not negligible, this effect is unlikely to be so large as to change the overall conclusions about the magnitude of the ILUC impacts estimated in chapters 4 to 8 of this report.

There are clearly risks of ILUC associated with biofuels. Attempts to quantify the impact, though uncertain, provide an indication of the risk, and more work needs to be done to work out how to reduce it. Through the use of ILUC scenarios, it has been possible to identify the type of mitigation actions that are needed to lower the risk of ILUC. Key actions that would lower the ILUC impacts calculated in this report include:

- effective global protection of high carbon stock land,
- use of low carbon stock areas for biofuel feedstock cultivation (although it is important to make sure that this land is not already used for other purposes such as pasture),
- above baseline yield increases (without a corresponding increase in nitrogen fertiliser use per tonne of output),
- improvements in supply chain efficiency,
- ensuring co-products from biofuel production are used as a replacement of land based products,
- integration of livestock and crop production systems.

As this study has not explored the feasibility of introducing mitigation actions or assessing their likely effectiveness, it is not possible to assess whether the ILUC risks associated with certain types of biofuels can be lowered. However, it is possible to see through this work the potential reduction in GHG emissions if some of these mitigation actions are effectively implemented. As the debate continues about the best way to deal with ILUC caused by biofuels, it is clear that there is still much work to be done in improving our understanding and modelling of how the future agricultural sector as a whole will evolve, the carbon stock changes associated with different land use changes in different geographic locations, and the extent to which LUC can be mitigated through local, regional and global efforts.

2 Introduction

The additional production of biomass feedstocks to replace products diverted to biofuel production from other markets may indirectly result in the expansion of agricultural land at the expense of other land uses. This is known as Indirect Land Use Change (ILUC), and has been raised as an issue of concern by recent scientific journal articles because of greenhouse gas (GHG) emissions and other environmental damage that it may cause (Searchinger et al., 2008; UNEP, 2009; FAO, 2008).

Policy makers have been considering ways in which the impacts of ILUC should be considered in policies related to biofuels. The United States Environmental Protection Agency (US EPA) has included an ILUC factor, i.e. a measure of CO_2e emissions associated with ILUC per unit of biofuel, in the calculation of the GHG balance of the different biofuel chains currently considered in the US (US EPA, 2010). The ILUC factors have been derived using a combination of two econometric models: the Forest and Agricultural Sector Optimisation Model (FASOM) and the Food and Agricultural Policy and Research Institute model maintained by the Center for Agricultural and Rural Development (FAPRI-CARD) (US EPA, 2010). However, there is much debate in scientific, policy-making and industrial circles around the validity of ILUC factors because of the associated scenario, modelling and scientific uncertainties (RFA, 2008a; Ecofys, 2009; Babcok, 2009; Mathews and Tan, 2009).

The European Commission has yet to decide on its approach to dealing with ILUC caused by biofuels (EC, 2009a) and recently published a consultation to receive feedback on the policy approach it should take (EC, 2010). Different options are being discussed, ranging from monitoring the impacts (through trends in certain key parameters to be determined) to different mitigation actions (e.g. international agreements on protecting carbon-rich habitats, extending use of bonuses/penalties to encourage or discourage the use of certain types of biofuels, etc.) or the inclusion of ILUC factors in the GHG emission saving calculations.

E4tech has been commissioned by the UK's Department for Transport (DfT) to contribute to the understanding of factors causing ILUC, management factors that can mitigate ILUC, and the magnitude of these effects. Specifically, E4tech has been asked, to develop ILUC factors for five different biofuel chains: bioethanol produced from wheat and sugarcane, and biodiesel produced from oilseed rape, palm and soy. The ILUC factors will be a measure of all the GHG emissions resulting from indirect land use change caused by a particular biofuel per unit of energy – i.e. g CO_2e / MJ. The results of this study are expected to contribute to an understanding of the risk of ILUC associated with different biofuels, of the potential magnitude of ILUC related effects, and of the uncertainties associated with understanding and quantifying ILUC effects. Clearly, there are other important issues closely interlinked with land use change, such as biodiversity protection and land rights but these are outside the scope of this study.

Current attempts to improve our understanding of the GHG impacts of ILUC focus on use of partial or general equilibrium models. While these approaches may ultimately be the most effective solution to assessing ILUC impacts, current models have weaknesses that need to be addressed. In this study, causal-descriptive modelling is used to develop ILUC factors. This approach uses cause and effect logic to describe the behaviour of a particular system, based on observations of how it functions. It provides a transparent analysis that enables input and review from stakeholders, and stakeholder participation has been a key component of this study. Importantly, the causal descriptive modelling can potentially serve as input to other modelling approaches.

The following part of this report is divided into 8 chapters. Chapter 3 defines the causal-descriptive model used, provides a discussion of the general approach, and presents the baseline and biofuels projections used in this study. It also provides a detailed description of the methodology. Chapters 4 to 8 then present the calculation of ILUC factors for each of the five chains studied based on the causal-descriptive approach. The biodiesel chains are presented first (palm, rapeseed and soy) followed by the bioethanol chains (wheat and sugarcane). Chapter 9 discusses in more detail key actions for mitigating ILUC, which are identified in the five previous chapters. Finally, chapter 10 provides a discussion of the outputs of this study and the lessons learned, and identifies areas for future work to improve the accuracy of analysis of this kind.

3 The causal-descriptive approach to indirect land use change

This chapter is dedicated to the description of the methodology used for calculating the ILUC factors presented in chapters 4 to 7. First, the causal-descriptive approach is defined. Then, in section 3.2, the baseline and biofuel projections are discussed in more detail. Section 3.3 then presents the core methodology, including the system definition and system boundaries, quantification of market responses, cause and effect relationships, and land and GHG impacts. Finally, section 3.4 discusses our approach to management practices and mitigation factors.

3.1 General approach

Modelling ILUC using causal-descriptive techniques requires mapping out all the impacts an increased demand for a certain biofuel has on the broader agricultural and land use systems – see Figure 1 below for a hypothetical example of the impacts of an above baseline demand for crop A due to its use in biofuel production. The aim of this process is to identify the possible land use changes that a biofuel can cause.

The land use change impacts of a certain biofuel can then be established by comparing the (worldwide) land use when the biofuel feedstock is produced to the (worldwide) land use with no additional demand for biofuels, i.e. the difference between the baseline and the biofuels projection. For the purpose of this study, the following two projections were defined:

- Baseline: no additional biofuels supplied from 2008;
- Biofuels projection: additional demand for biofuels based on worldwide government targets.

To quantify the land used in each of these projections, we have used a combination of four different approaches:

1. **Statistical analysis of historical trends** was used to quantify the market responses to the additional feedstock demand and estimate business as usual trends.
2. **Market analysis** was used to gain insights into likely evolution of markets (such as the entrance of new products or the creation of new markets) and to identify product substitutions. When necessary, the projections obtained through extrapolation of historic trends have been adapted to take the results of the market analysis into account.
3. **Expert input and literature review** provided qualitative validation of the results of the statistical and economic analyses.
4. **Variations in parameters** from the statistical analysis to reflect different potentially likely ILUC scenarios.

Therefore, unlike the equilibrium models used by others in attempting to model ILUC impacts, land and commodity prices are not explicitly considered or modelled. However, prices are implicitly considered by using projections based on historical trends and expert opinion to understand deviation from historic trends, for example, which country is likely to be the marginal exporter of a particular commodity in 2020. Where possible, we have attempted to record the reasoning and rationale behind these views.

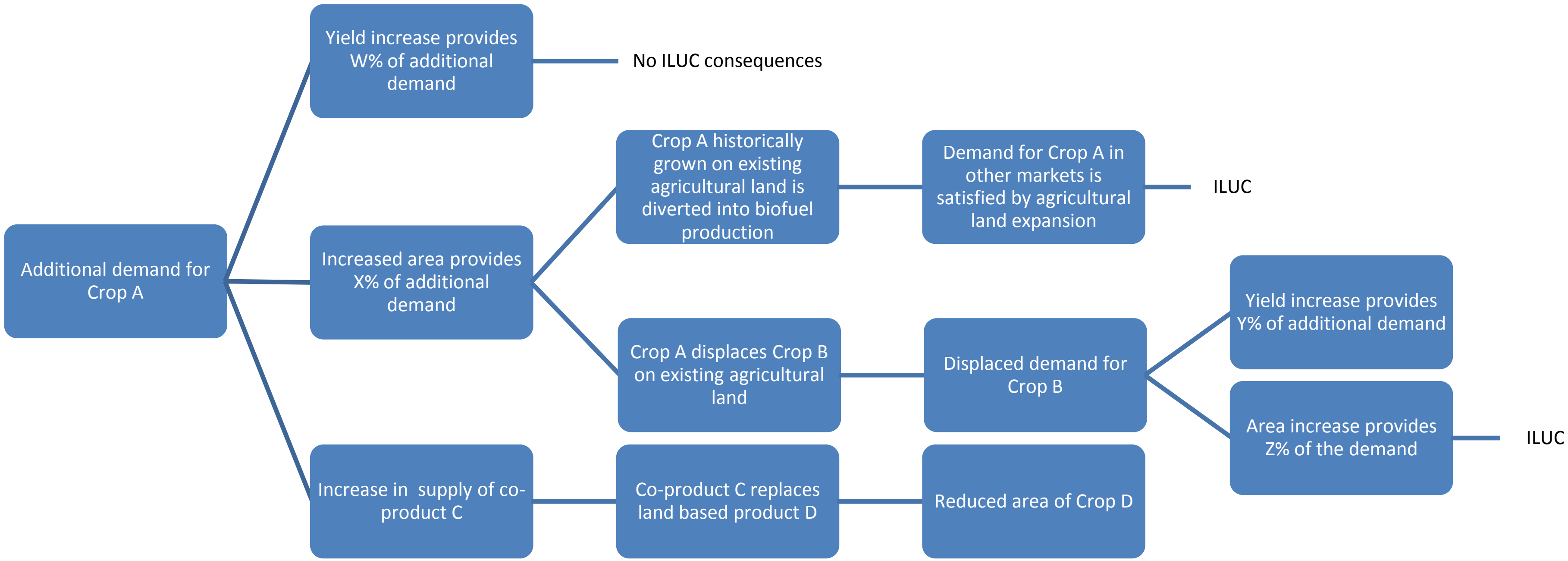

Figure 1. Example of a causal-descriptive approach to ILUC quantification.

In this example, the additional demand for crop A is met through two main market responses: an increase in yield associated with no ILUC consequences and an expansion in cultivation area for biofuel production. In the first case, crop A is grown on the same land "as usual" but is diverted from a historical market into biofuel production, thus leading to agricultural land expansion to satisfy the demand for crop A in its historical market. In the other case, the agricultural land expansion reduces cultivation area for crop B, which then has to be produced in some other way. In this example, crop B is now produced through an increase in yields and another area expansion. Each area expansion ultimately leads to ILUC impacts. Furthermore, the increased production of biofuel from crop A leads to the production of co-product C which replaces another land-based product (crop D) and thus "saves" some land.

Authoritative data sources were used whenever possible to ensure the robustness of the study. Data has been referenced at each step of the analysis and is discussed throughout the report. Three main data sources can be identified:

- FAOSTAT, database maintained by The Food and Agricultural Organisation of the United Nations (FAO), especially the databases on production and trade;
- The 2009 U.S. and World Agricultural Outlook Database maintained by the Food and Agricultural Policy Research Institute (FAPRI);
- The Production, Supply and Distribution (PSD) Online database maintained by the United States Department of Agriculture – Foreign Agricultural Service (USDA FAS).

Stakeholder involvement was a crucial part of this project, especially as the causal-descriptive approach we have adopted allows for transparency in the analysis. Input from stakeholders was gathered through three different mechanisms:

- An Expert Advisory Group (EAG) provided regular input to the study, reviewed all aspects of the approach and commented on them. Members of the EAG were selected based on their technical expertise which covers areas such as specific biofuel chains, agricultural and related markets, approaches to ILUC impact modelling, etc.
- Three stakeholder meetings were held during the course of the project. The first was aimed at gathering views and insights on the causal-descriptive approach while the two others focused on chain modelling and results. The meetings were open to all stakeholders and invitations were sent to industry, government, non-government organisations and to other experts.
- During the course of the project, a website enabled active communication between E4tech and the stakeholders and extended the reach of the project. Through the website and other announcements, stakeholders were also invited to submit comments and evidence by email.

Finally, ILUC modelling is complex and uncertain for several reasons. First, it is about projecting impacts in the future, which is inherently uncertain. Second, ILUC models cannot be validated or calibrated against historic data: indirect land use change is not an observable parameter, as so it has not been measured historically. This means that high uncertainty and debate exist around the exact cause and effect relationships that lead to land use change. Thirdly, there are uncertainties associated with the estimates of carbon stocks associated with different land types, and the carbon stock losses associated with land use change.

To take into consideration uncertainties around future ILUC impacts, we did not calculate a single ILUC factor for each of the five chains examined, but we developed a series of scenarios in which assumptions and parameters have been varied to reflect potential future situations and assess their influence on the results. Uncertainties associated with carbon stocks have been included as an ILUC factor error bar for each scenario.

3.2 Baseline and biofuel projections

The baseline and biofuel projections provide the context within which the ILUC impacts are assessed, and as such they are at the core of a causal-descriptive approach to calculating ILUC factors. Although this project is limited to five biofuel chains, the baseline and biofuel projections must consider demand for all crops that result in a demand for land in 2020. This project considers demand for the most important (i.e. most cultivated) crops, with special attention to the crops for which demand is likely to change as a results of demand for biofuels. The projections therefore consider the demand for food, feed, fibre and fuel crops in 2020, but differ in the assumptions on biofuel demand.

Both projections have the same starting point: demand for crops and land for food, feed, fibre and fuel in 2008 and both project demand for these crops until 2020. In the baseline, the demand for biofuel remains at the same level as in 2008. In the biofuel projection, the demand for biofuels grows to meet worldwide targets for biofuels in 2020. The demand for crops for food, feed and fibre to 2020 is the same in both projections.

As such, the baseline is not demand in 2008 but the projection of demand for crops to 2020, when demand for biofuel is held constant at its 2008 level. The projection of demand for food, feed and fibre are based on projections by FAPRI. FAPRI does not base its projections on the starting year alone but on historical trends under average weather patterns, taking into account existing farm policy and policy commitments under current trade agreements and custom unions (FAPRI, 2009b). This means that even if 2008 was a peculiar year in terms of commodity prices and production, projections were not based on 2008 alone.

Furthermore, to calculate the ILUC factors, the land use in 2020 in both the baseline and the biofuel projections were compared and the difference between the two projections was assigned to biofuels. Thus the choice of the starting year of the projections (i.e. 2008) only has a minimal effect on the estimated ILUC factors. The following sub-sections describe how each of the projections was built.

3.2.1 The baseline

The baseline is based largely on FAPRI (2009a) projections, as there was a general consensus in the EAG that they provide a valid short-term projection for agricultural commodities. However, while FAPRI's projections incorporate increased biofuel feedstock production from 2008 to 2018, we have held production of biofuels constant in absolute terms between 2008 and 2018 in the baseline. Also, FAPRI's 2009 projections only go out as far as 2018, so the projections were extended to 2020 by extrapolating the 2008 to 2018 compound annual growth rate (CAGR) in production (excluding biofuel feedstock production) out to 2020.

However, for oilseed rape demand in Europe, the FAPRI projections to 2018 were not used to quantify our baseline demand in 2020. FAPRI projects that demand for oilseed rape (after taking the demand for biodiesel production out) would reach 20.7 million tonnes in 2018, i.e. an increase of 1.6 million tonnes just for food and exports. Such an increase was felt unrealistic in light of other data sources such as the USDA FAS database. In this database, it is possible to distinguish the demand for different uses (domestic food consumption, domestic feed consumption, industrial domestic consumption, exports, etc.). If we use the industrial domestic consumption as a proxy for biodiesel production, we can see that after a period of rapid growth in food consumption and decline in exports (in the 1990's), European consumption of rapeseed oil has stabilised in the years 2000 (graphs showing these trends are included in Annex 1). It was therefore decided to keep the demand for oilseed rape in Europe constant between 2008 and 2020.

If FAPRI projections had been used, Europe would have produced a lower proportion of its own demand for oilseed rape and thus would have imported more of it. Oilseed rape ILUC scenario 6 (see chapter 5) examines such a case and can thus show the effect on the ILUC factor of assuming an increase in consumption of oilseed rape for food and feed.

Only one baseline has been used in this study. An interesting area for further work would be to look at alternative baselines and explore the impacts that they have on the ILUC impacts calculated. However, as described above, some of the scenarios developed do result in the same types of effects as if a different baseline had been used.

The aim has been to calculate the ILUC impacts associated with producing a MJ of a certain type of biofuel. Because palm and soybean are traded internationally, it was considered that the ILUC impacts of a MJ of these biofuels would be the same anywhere in the world. However, it was considered that there would only be significant demand for wheat ethanol and rapeseed biodiesel in Europe and therefore the ILUC impacts would be driven by the demand for these types of biofuels in Europe. As such we have calculated ILUC factors for soybean and palm oil biodiesel based on the global demand for these commodities for biofuel, and ILUC factors for wheat ethanol and rapeseed biodiesel based on European demand for these commodities for biofuel.

Table 1 below shows the baseline projections for the most important commodities and for the regional scale considered. Commodities used during the modelling will be discussed in more detail in the relevant sections of the following chapters.

Table 1. Projections of the production of different agricultural commodities in the baseline. Based on FAPRI (2009a) and USDA FAS (2010).

Feedstock	Region	Production ['000 tonnes]	
		2008	2020
Barley	Europe	65,579	62,619
Corn	Europe	61,197	60,901
Oilseed rape	Europe	19,100	19,100
Oilseed rape	North America	13,255	14,397
Palm fruit	Worldwide	208,353	298,751
Soybean	Worldwide	230,286	304,562
Sugarcane	Worldwide	1,702,270	1,988,666
Wheat	Europe	150,514	147,962

3.2.2 The biofuel projection

The projected demand for biofuels in 2020 is estimated by assuming certain biofuel targets are met in each world region. These targets are shown in Table 2.

Table 2. Biofuel targets by world region which are assumed to be met in the biofuel projection.

Region	% ethanol in gasoline	% biodiesel in diesel
OECD NA	10	3
EU	10	9
OECD Pacific	7	4
Eastern Europe	0	0
China	4	1
Other Asia	5	3
India	7	6
Middle East	0	0
Latin America	34	4
Africa	2	1

The total assumed consumption of biofuel in 2020 (144bn litres bioethanol and 46bn litres biodiesel) is in line with the projections in the 450 scenario from the 2009 World Energy Outlook (WEO) (IEA, 2009).

The biofuel projection to 2020 does not include the relatively small amount that would come from 2nd generation (2G) bioethanol and biodiesel, as we assumed that, in the short term, this would mainly come from wastes and residues rather than land based energy crops. However, this is certainly a sensitivity that could be explored in further work.

We have based the split between biodiesel feedstocks in different regions on previous analysis carried out for the Gallagher Review (E4tech, 2008). In Europe, this split was revised based on a scenario put forward in our stakeholder meetings – which provides a breakdown between feedstocks used for FAME and HVO and the split between these two types of biodiesel.

The split between bioethanol feedstocks was also based on E4tech (2008) and updated to better reflect EAG and stakeholder views. For example, in E4tech (2008), Europe was assumed not to use any corn for bioethanol production. In this study however, we have considered that ~22% of EU bioethanol will be produced from corn.

Feedstock supply constraint due to land availability for cultivation area expansion due to biofuels in 2020 was explicitly considered in this study. Other constraints, such as the European Renewable Energy Directive sustainability criteria, were not explicitly modelled.

The assumed contributions of the different feedstocks to biofuel production in 2020 in the different world regions are presented in Table 3 below. Based on these contributions, we determine the total amount of biodiesel and bioethanol projected to be produced from a certain feedstock. Then, we used the conversion ratios from the Renewable Transport Fuel Obligation (RTFO) scheme (RFA, 2009) to calculate the demand for the different feedstocks in all the world regions in 2020 for biofuel production[1]. Table 4 below summarises this additional demand due to biofuels for the five biofuel feedstock studied.

Table 3. Contributions of different feedstocks to the biodiesel and bioethanol production in 2020 in our biofuel projection.

Region	Contribution to 1G bioethanol production [%]						Contribution to 1G biodiesel production [%]				
	Cassava	Wheat	Sugar beet	Sugarcane + Molasses	Sorghum	Maize	Soybean	Palm	Sunflower	Jatropha	Oilseed rape
Africa	20	-	-	80	-	-	15	50	-	20	15
China	10	28	-	17	17	28	33	33	-	-	33
Eastern Europe	-	-	-	-	-	-	-	-	-	-	-
European Union	-	67	1	10	-	22	27	24	-	5	41
India	20	-	-	80	-	-	25	-	-	50	25
Latin America	5	-	-	95	-	-	85	15	-	-	-
Middle East	-	-	-	-	-	-	-	-	-	-	-
OCED North America	-	-	-	30	-	70	72	25	-	-	3
OECD Pacific	-	25	-	75	-	-	13	7	5	-	75
Other Asia	20	-	-	80	-	-	-	100	-	-	-

As can be seen in Table 3, although this study concentrates on five feedstocks, the demand for other biofuel feedstocks was taken into account. For example, demand in the US for biofuel feedstocks such as corn, wheat and sugarcane is considered in this study. Furthermore, the impact of corn DDGS production due to corn bioethanol production on wheat demand in Europe and soybean meal demand was also accounted for. However, this study did not explicitly model the land use change caused by the increased demand for corn for bioethanol in the US. We do not expect this to influence the results of any of the other crops, considering the potential for corn cultivation expansion in the US and the low US supply of the studied biofuel feedstocks.

[1] In the case of palm oil extraction from fresh fruit bunches (FFB), stakeholders judged the RTFO conversion factor too low (0.16 t oil / t FFB). Instead a value of 0.20 t oil / t FFB was used.

Table 4. Demand for the feedstocks examined in this study (i.e. oilseed rape, palm, soybean, wheat and sugarcane) in the biofuel projection in 2008 and 2020.
Note: demand is for all uses including biofuels.

Biofuel feedstock	Region	Demand [1000 tonnes]	
		In 2008	In 2020
Oilseed rape	Europe	19,100	26,264
Palm fruit	Worldwide	208,353	381,645
Soybean	Worldwide	230,286	365,288
Wheat	Europe	150,514	174,455
Sugarcane[2]	Worldwide	1,702,270	2,373,310

As in the baseline, we have only developed one biofuel demand and supply projection in most cases. For sugarcane ethanol, we do explore the impact of an alternative biofuel projection, in which there is a much higher demand for sugarcane (see section 8.5 for a view on the impacts of this). In the case of sugarcane, the ILUC impacts of increasing demand are heavily dependent on the assumption around where that increasing supply comes from. If the additional production is assumed to come from exactly the same land types as for a lower biofuel projection, the ILUC impacts per MJ biofuel will be the same. If the additional production is assumed to come from land with lower ILUC consequences or carbon stocks than for the lower projection, the ILUC impacts (per MJ of fuel) will be lower. Similarly, if they come from land with higher ILUC consequences the ILUC impacts per MJ biofuel will be higher.

3.3 Methodology

Section 3.1 gave a general overview of the causal-descriptive approach to modelling ILUC and section 3.2 discussed the baseline and biofuel projections. In this section, we discuss in more detail the methodology for estimating first the market responses to an additional demand for crops for biofuel production and an additional supply of co-products from biofuel production, and then the cause and effect relationships that link these market responses to ILUC impacts.

We start with a description of the system examined and the definition of the study boundaries. Then we look at how to quantify the market responses and cause and effect relationships due to the additional demand for biofuel feedstock (section 3.3.2) and to the increased supply of biofuel co-products (section 3.3.3). In section 3.3.4, we explain how we have calculated the GHG emissions from the land use changes identified in the previous sections. Finally, section 3.3.4.2 presents how we have dealt with uncertainty.

[2] As discussed in section 8.2.1 of this report, ethanol can be made from sugarcane juice or molasses (a co-product of sugar production). Here we only include the ethanol produced from sugarcane juice ethanol because the direct and indirect impacts of using molasses ethanol are very different for the two feedstocks and so need to be considered separately. In other words, the GHG impacts of using 1 MJ molasses ethanol are different to the GHG impacts of using 1 MJ of sugarcane juice ethanol.

3.3.1 System description and boundaries

System description. Based on a literature review and discussion with experts, six key market responses were identified as ways in which the additional demand for biofuel feedstock in a certain world region can be satisfied. These are:

1. Increased production area of the biofuel feedstock;
2. Above baseline increase in yields of the biofuel feedstock;
3. Substitution of the biofuel feedstock in other markets by other suitable products;
4. Changes in the trade balance of the biofuel feedstock;
5. Increased availability through improvements in efficiency in the supply chain of the biofuel feedstock;
6. Reduction in demand for the biofuel feedstock in other markets, due to increased prices.

Figure 2 below gives an overview of how these market responses relate to land use change. In this figure, the two first market responses have been linked under the title "increased production of Crop A". Indeed, yield and area response to additional demand are closely linked and they will be projected to 2020 using an approach discussed in a later section (see section 3.3.2.2).

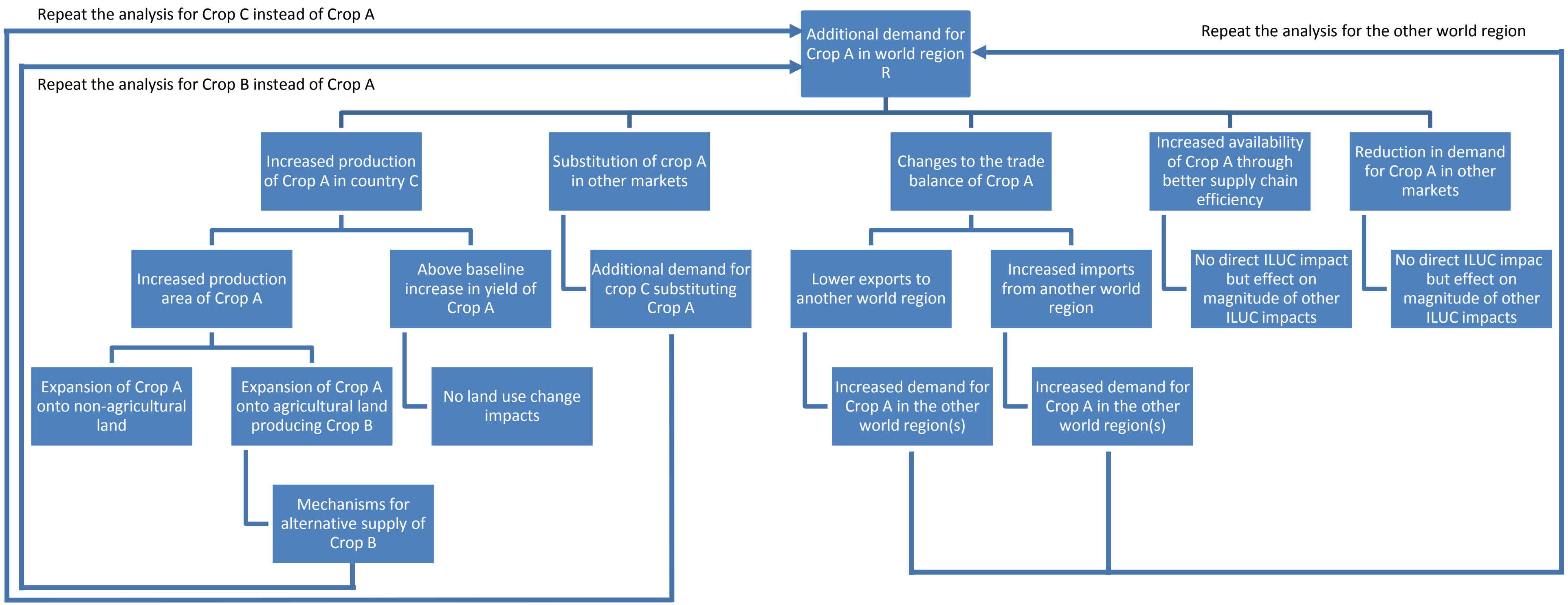

Figure 2. Overview of the cause and effect relations that can lead the market responses to cause land use change.

Substitution. One way of satisfying the demand for the biofuel feedstock is to displace the feedstock out of its current use / market and into biofuel production. This can either happen domestically, and then another product will have to substitute for the no-longer supplied biofuel feedstock in the traditional domestic market (market response 3); or it can have consequences in other world regions, when the biofuel feedstock is diverted out of exports or when more feedstock is imported to the world region with the increased demand (market response 4). These market responses have land use impacts as an increased amount of either the substituting product or the feedstock itself now has to be produced (domestically or in another world region).

Impacts of co-products. The increased production of biofuels actually has a seventh market impact: it leads to increased production of co-products, some of which displace other land-based products. In the approach followed in this analysis, the biofuel gets a credit equal to the amount of GHG saved by the LUC impacts avoided as a result of biofuel co-products displacing land-based product(s).

As for the ILUC impact due to additional demand for agricultural products (cf. section 3.2), we did not consider any change in overall food demand and production due to the additional production of biofuel co-products. For example, many of the biofuel co-products can be used as animal fodder; we did not consider that this would lead to increased meat production in the biofuel projection compared to the baseline. We only considered that the co-products would lead to lower production of other animal fodder types.

Figure 3 shows how the increased supply of co-products can lead to (avoided) land use change. If biofuel co-products displace other land-based products, the lower demand for the latter can induce several market responses (lower domestic production, change in trade balance or displacement of other products in other markets). These lead to land use changes either in the same world region or in another through the same mechanisms as shown in Figure 2.

As shown in Figure 3, some of the displaced product(s) may be produced together with other co-product(s). If the displaced product is the determining product[3], these co-products will no longer be produced, leaving a gap in the demand that should be filled by another product. Thus biofuel co-products, even though they displace some products, can lead to an increased demand for other products. If the displaced product is not the determining product (i.e. is a dependant co-product), it should be studied whether the increased supply of biofuel co-product is causing enough change in the demand for the dependent product to actually affect production volume (and thus the production of the determinant co-product) or whether the change in demand will only lead to the dependent co-product being supplied to another market or discarded.

[3] A determining co-product is a co-product that determines the production volume of the process. The other co-products are called dependent. Determining and dependent products can change over time, depending on parameters such as price and demand.

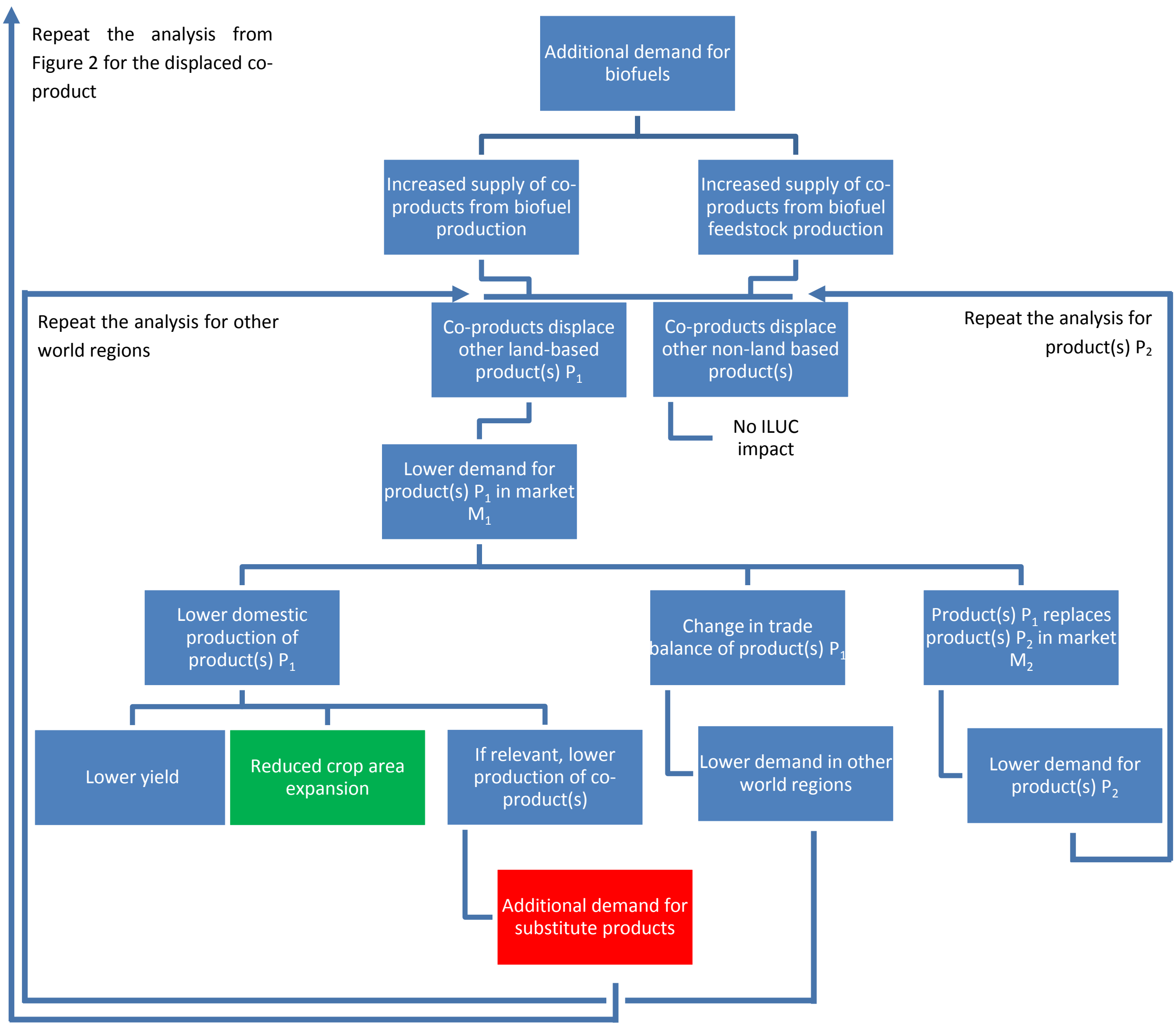

Figure 3. Overview of the cause and effect relations that can lead the additional supply of co-product to cause or avoid land use change.

The green box indicates that, if this box is reached, the biofuel will get a credit for avoided area expansion. On the other hand, the red box indicates that, if it is reached, the biofuel will get a debit as it is causing increased demand for other products.

Boundaries. Even though identified as a possible market response to increasing demand, lack of evidence on supply chain efficiency improvements makes the assessment of market response 5 difficult. Furthermore, the magnitude of this market response is likely not to be important in terms of ILUC in the period to 2020. Thus market response 5 has been excluded from this study. We recognise this is an uncertainty factor that should be investigated in further work. This market response will, however, be considered when discussing management practices and ILUC mitigation factors (see chapter 9).

Increasing demand for biofuel feedstocks is likely to lead, at least in the short-term, to higher prices which will impact the demand for this same feedstock in other markets (i.e. market response 6 above). It has for example been identified that increasing biofuel demand, among other causes, recently led to increasing food prices which impacted the demand for food, especially in the poorer fractions of the world populations. Such impacts are undesirable consequences of biofuels, so market response 6 was not considered further in this study and the ILUC impacts calculated may be considered conservative as a consequence.

Furthermore, this study concentrated on evaluating the indirect impacts of increased demand for biofuels on land use. However, there may be other indirect effects, i.e. not land related, which could also be attributed to the increase in biofuel demand. For example, increases in yields in response to higher biofuel demand may be achieved through increased nitrogen fertiliser application, with consequences on the GHG emissions of the feedstock cultivation (both for biofuel production and for other uses of the crop). These emissions are not included in the results presented in chapters 4 to 8. This issue is discussed in more detail in section 3.3.4.2.

Such non-land related indirect effects can potentially be quite important compared to the direct chain related GHG emissions of biofuels. However, and unless otherwise clearly stated in this report, the GHG impacts of these non-land related indirect effects have not been taken into account.

Avoiding double counting. Use of ILUC factors is one of the possible policies aimed at mitigating LUC. Other mechanisms include specific land type protection policies (e.g. the United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (UN-REDD)) or biodiversity protection programmes. These different policies create a risk of double-counting the credits, as these can be attributed to both the biofuel's ILUC factor and to the land use change avoidance policy. It is therefore important to have an accurate baseline for including land savings from co-products. For example, if a co-product is credited with avoided deforestation, it would be important to make sure that the effectiveness of external measures aimed at avoiding deforestation, such as through REDD, are included in the baseline. This is important for ensuring that the ILUC factor only credits avoided deforestation that is actually happening.

3.3.2 Cause and effect relationships of market responses

Assessing the magnitude of the market responses to an increase in the demand for a biofuel feedstock requires working out a relationship between the change in demand and the different responses. The principal responses we are interested in to understand ILUC impacts are how yields and agricultural areas vary and what product substitutions occur as the result of changes in demand for crops under the biofuels projection.

Such relationships can be built based on prices, directly on demand based on historic trends, or through an analysis of product properties and markets.

Whether prices should be used has been widely discussed in the context of this project – with no clear consensus appearing. Partial or global equilibrium models have mostly used price elasticities to model the various responses leading to ILUC: for example, modelling done in the Global Trade Analysis Project (GTAP) and by Searchinger et al. (2008) is based on price elasticities (Ecofys, 2009; Babcock, 2010). However, price-based analyses have their limitations:

- **Demand is not the only factor affecting commodity price.** The recent large increases in world crop prices have been primarily caused by poor harvests (thus lower production), spikes in other commodities (e.g. oil) and faster than expected increases in consumption

(JRC, 2008). Furthermore, the price influences are quite different for different crops (RFA, 2008b). It is thus difficult to determine the exact relationship between demand and price.

- **Price alone is not sufficient to increase yields.** Hazell (2008) suggests that concerted investment in R&D is also required. Furthermore, from the consultations ADAS undertook for the Gallagher review (RFA, 2008a), it also appeared that advances in productivity depend on a combination of three drivers: (i) public investment in research and infrastructure, (ii) supportive legislative and trade agreements, and (iii) private investment supported by profitability of production.
- **The effect of prices takes several years to be reflected in yield and land use changes.** Increased output as a result of higher prices is in large part due to longer term investment in cultivating new land, land improvements, machinery, infrastructure and higher yielding crops for the years following high prices. Thus the effect of prices takes several years to be fully reflected in yield and land use changes.

In this study we have decided not to use price elasticities, but to analyse the market responses through direct demand-based relationships for yields based on historic trends and through product properties and market analysis for product substitution. However, we have also used expert opinion to understand if extrapolations of historic trends are realistic based on their understanding of the particular markets studied. Also, prices are implicitly considered by using projections based on historical trends and expert opinion to understand deviation from historic trends, for example, which country is likely to be the marginal exporter of a particular commodity in 2020. Where possible, we have included the rationale behind such assumptions. The experts' views on how markets are likely to evolve in the future have also played a role in deciding which alternative future scenarios should be studied.

The following sub-sections explain in more detail how the market responses and their cause-effect relationships were quantified.

3.3.2.1 Product substitution

For a product to be considered relevant as a potential product substitute, it must be seen to be fulfilling the same needs in a certain market. This can ultimately be expressed in terms of properties of the product. Product properties may be divided in three groups depending on their importance (Weidema, 2003):

- **Obligatory properties** that the product must have in order to be at all considered as a relevant alternative;
- **Positioning properties** that are considered nice to have and which may therefore position the product more favourably relative to other products with the same obligatory properties;
- **Market-irrelevant properties** that do not play a role for the customer's preferences.

This approach helps to gain knowledge on the possible products that can substitute for a feedstock that has been diverted to the biofuels market (or on the possible products a biofuel co-product can displace). Analysis of historical trends in market shares of the biofuel feedstock and the possible substituting products in their traditional markets can be used to identify where product substitution has taken place historically, in periods of changes in demand and supply.

However, as the markets and the products present on the different markets, together with the importance and relevance of product properties, change over time, we systematically submitted the conclusions from our analysis of historical trends to the EAG for approval and compared them to results in literature. This allowed us to take into account questions such as whether there are any political or policy constraints that could prevent the substitution.

If product substitution occurs, the impact of the additional demand of substituting product(s) needs to be quantified. First, a substitution ratio between the biofuel feedstock and the substituting product(s) needs to be determined. These ratios were usually taken from literature and cross checked with experts. This enabled the calculation of the amount of additional demand for the substituting product that is created. To determine the land use impacts of this additional demand, we then need to perform the analysis as shown on Figure 2.

Table 5 below summarises the main consequential questions that can influence the result of the analysis, and the approach we have taken to answer them.

Table 5. Summary of the consequential questions and approaches taken to determine the contribution of product substitution to the land use change impact of biofuels.
The biofuel feedstock is referred to as Crop A in this table.

Question	Approach	Description
What products substitute for Crop A in non-biofuel markets?	Market analysis	Identify the traditional markets for Crop A and the potential products that could substitute Crop A on these market.
	Expert inputs and literature review	• Have all the current uses for Crop A in non-biofuel markets been identified?
How much of the additional demand results in other products substituting for Crop A in non-biofuel markets?	Statistical analysis of historical trends	Analysis of historical trends in imports, consumption and prices, to estimate the amount of substitution taking place due to an increased demand for Crop A in the biofuel market.

	Expert input and literature review	Can the product substitution effects suggested by statistical analysis be achieved?
		• Are the suggested substitutions likely to take place considering the technical compatibility, availability of supply and relative economics?
		• Are there any political or policy constraints that would prevent this substitution?
		• Are there any examples to illustrate that these product substitutions occur?
		• Are the substituting products likely to change over time?
		  • Are there constraints in the traditional markets for Crop A?
		  • What is the future evolution of these markets likely to be?

What is the additional demand for the substituting product(s)?	Expert inputs and literature review	Estimate the substitution ratio between Crop A and the other product. • Is there consensus on the technical equivalence of Crop A and its substitute product(s)? • Review of the substitution ratio(s) – are they in an acceptable range?

Question	Approach	Description
How is this additional production supplied?	Repeat method to assess how demand for displacing product is met	A separate analysis to establish the impact on yield, area, substitution and co-products for the affected product system(s) is required. This can be carried out using the same framework applied to biofuel feedstocks. The same rules on boundaries as described in section 3.3.1 apply.

The substitution effect does not always need to be direct. For example, when soy oil is used for biofuel, palm oil may ultimately be the crop whose production is expanded, as it is considered to be the marginal crop. However, soy oil may not be being directly displaced by palm oil, there may be a chain of substitutions in different markets which lead to the palm oil expansion.

3.3.2.2 Yield and area changes

Marginal vs. average yield. Determining the yield of the crop in 2020 on a particular area of land is essential for quantifying the area expansion needed to satisfy the demand for that crop. However, yields depend on many factors including the suitability of the land to the cultivation of a certain feedstock. Under the assumption that all the well-suited land has already been taken into production, it has been argued that if agricultural area expansion occurs, it would be on less suitable land and thus have lower associated yields than for the land already in production. For example, the analysis for the Californian Air Resource Board (CARB) by the GTAP model assumed a ratio between yields on newly converted land and yields on existing cropland (for the same crop) in the range of 0.5 to 0.75. Sensitivity analysis indicates that a change from 0.5 to 0.75 results in a 38% reduction in LUC intensity (Babcock and Carriquiry, 2010). Edwards (2010) also pointed out that marginal yields may be much lower than average yields. Using the marginal approach, we come to a situation in which the marginal yield is the yield achieved on the poorest quality land of the least productive farmers in a particular region, i.e. it is very low.

These assumptions (both the fact that all well-suited land is already in production and that marginal yield is lower than average yield) have been widely discussed. Utilising agro-ecological zoning and land cover information, Fischer et al. (2002) estimated that close to 19% of the global land with rain-fed cultivation potential was under forest ecosystems. For Western Europe, agricultural land in 1994-1996 was 35.1 million hectares whereas well-suited land was estimated at 64.2 million hectares. Others (e.g. Campbell et al., 2008) have argued that a significant amount of abandoned agricultural land exists worldwide. This land was once considered suitable for agricultural production and could become suitable again.

Furthermore, Babcock and Carriquiry (2010) looked into the assumption that yields on newly converted land are lower than on existing cropland. They concluded that this assumption and the size of the difference very much depends on the world region (e.g. assumption may be correct in the United States but not in Brazil) and on the crops considered.

Besides, marginal yields, like any marginal data, are well suited when small changes are analysed that do not lead to a systemic change in, in the case of yields, the type of varieties grown or management practices. This is because marginal data only incorporates changes to one parameter, such as soil quality for example. It is however very difficult to develop marginal data that would incorporate changes to several parameters at one time. In this study, we examine the differences between two different systems: one with no additional demand for biofuels and one with additional demand. As can be seen when comparing Table 1 and Table 4, the changes in demand between the two systems cannot be considered small relative to the total demand. Such changes in demand will induce systemic changes (for example, we are likely to see more high-yielding wheat varieties being grown for bioethanol production, agricultural practices will tend to increase yield overall because of substantially increased demand, etc.) that cannot be capture by using marginal yields.

Average data, and especially average yields, were thus considered more suitable for the quantification of the indirect land use change impacts of biofuels and are thus used in this study. In the rest of the report, we will simply use the term “yield” to refer to “average yield”.

Yield and area projections. Determination of the GHG emissions from land conversion requires an assessment of the incremental agricultural land area that is required to meet the incremental demand for feedstock crops. This requires an evaluation of the shares of incremental output growth that will be met from yield improvements and area expansion. Lywood et al. (2009a) proposes an approach for calculating these shares of incremental growth for several crops in different world regions.

Lywood et al. (2009a) established direct relationships between historic changes in yield and land area for different crop-region combinations. These relationships were then used to determine the relative contribution of yield and area changes to output growth. Based on these “historical” relationships, two parameters were derived that enable the calculation of yield and area changes in the future when one knows the change in demand.

For the crop-region combinations studied in the paper, Lywood et al. (2009a) also showed that this approach provided better projections of historical trends than would do the approach through which the yield change is based on extrapolation of simple historic trend its average growth rate.

Based on these arguments, we decided to use the approach by Lywood et al. (2009a) for our yield and area projections (both in the baseline and in the biofuel projection).

The possibility of reaching the projected contributions of yield and area were then assessed using expert opinions and literature review. In case this sense check showed the projections to be unrealistic, these projections were revised to better fit the conclusions from the sense check. In these cases, the reasons for such a revision have been described in detail in the relevant sections of the specific fuel chain chapters.

Land use change due to feedstock cultivation area expansion. The impact of agricultural land expansion depends on many parameters:

- If the cultivation area is actually decreasing in the biofuels projection (but decreasing less than in the baseline), then land is becoming available for biofuel expansion. The impact of the lower decrease is the LUC from what would have happened on that land in the baseline. This could have been other agricultural production (then biofuel feedstock production results in a displacement of this agricultural production to other land leading to increased conversion of “natural” land to agricultural land or avoided reversion of agricultural land back to “natural” land) or reversion of the land back to its natural state (then biofuel feedstock production displaces carbon accumulation in the natural carbon sinks).

- If the cultivation area is actually expanding, then biofuel feedstock production displaces either agricultural production of other commodities or expands onto land not previously used for agriculture.

In both cases, the method is to follow the series of land use changes that ultimately results in either a reversion of the land back to its natural state or an agricultural expansion onto land not previously used for agricultural use. This is what is referred to in this report as a knock-on effect. We consider that the change of agricultural land from one annual crop cultivation to another does not lead to GHG emissions or loss of carbon sink. However, changes from “natural” land to agricultural land or from perennial to annual cropping (or vice-versa) lead to changes in carbon stocks and emission or uptake of GHGs.

Another important parameter to assess GHG impacts of ILUC is the actual area of land for which use changes. For example, if wheat expands onto one hectare of land used to grow another cereal crop in the baseline, this other crop then needs to grow somewhere else (this is the knock-on effect). How many hectares will it take to achieve the same production of this other cereal as in the baseline? This depends on agricultural yields that can be achieved on different types of land. In this study, we use average yields, because we examine system-level changes, i.e. we consider that the large increase in biofuel demand by 2020 will lead to wide changes to the agricultural system. The use of average yields leads us to consider that the production of a certain crop will take up the exact same amount of land, whether in the baseline or when displaced due to biofuel feedstock production. Thus the land area does not change throughout the knock-on chain. This view was supported by the recent literature review on ILUC performed for DG Energy which suggests that there is no evidence of yields being different on newly converted lands (DG Energy, 2010).

Table 6 summarises these possibilities and the approaches taken to quantify them.

Table 6. Summary of the consequential questions and approaches taken to determine the contribution of increased production to the land use change impact of biofuels.
The biofuel feedstock is referred to as Crop A in this table.

Question	Approach	Description
How much of the additional demand is met through yield improvement and area expansion of Crop A?	Statistical analysis of historical trends	The approach described in Lywood et al (2009a) is applied to assess the changes to yield and cultivation area due to increased demand for biofuels. The results are then sense checked through analysis of yield evolutions and land availability.
	Expert input and literature review	Are the projected yields and cultivation area realistic?
How much additional land is needed to produce the additional demand for biofuels?	Statistical analysis of historical trends and expert input	Based on the extrapolation of results of the statistical analysis, the changes in cultivation area of Crop A, both between 2008 and 2020 and between the two projections can be calculated. Determine the use the land on which Crop A is grown in the biofuel scenario would have had in the baseline.
If the previous land use was agricultural, where will the displaced product be produced?	Statistical analysis of historical trends	Review trends in agricultural production of the displaced products in the countries / regions where additional production occurs.
	Expert input and literature review	Are the projected trends realistic?

3.3.2.3 Changes in the trade balance of commodities

Based on the results of the two previous sub-sections (Product substitution and Yield and area changes) and on expert input and literature review, several scenarios in each of the studied biofuel chains will examine the impact of changes in import and export of products on the ILUC factors. Through this approach, we can investigate the difference in terms of ILUC between high domestic production of certain feedstocks and high international trade.

3.3.3 Increased co-product supply

The other market response to consider is the impact of the additional supply of co-products from the feedstock and biofuel production. Figure 3 on page 17 shows how an additional supply of co-product can lead to land use impacts.

There is a wide range of literature on biofuel co-products, the type of product they substitute and the substitution ratios. We have drawn on this literature, and the product property approach explained in section 3.3.2.1 to identify which products are displaced by the biofuel co-products and their quantities.

To assess the land impact avoided by the displacement of these products, we will use a combination of the approaches outlined in Figure 2 and Figure 3 and explained in more detail in the previous paragraphs. Table 7 below summarises these approaches and the consequential questions they address.

Table 7. Summary of the consequential questions and approaches taken to determine the contribution of increased supply of biofuel co-products to the land use change impact of biofuels.
The biofuel co-product is referred to as Co-product Z in this table.

Question	Approach	Description
What products will Co-product Z displace?	Market analysis	Identify the traditional market(s) for Co-product Z and the marginal product(s) that Co-product Z is likely to displace, taking into account technical and economic considerations.
	Expert inputs and literature review	• Have all the possible uses for Co-product Z been identified? • Are the suggested displacements likely to take place considering technical compatibility, availability of supply and relative economics?
What quantity of land-based products is displaced by Co-product Z?	Expert inputs and literature review	Estimate the substitution ratio between Co-product Z and the substituted product(s). • Is there consensus on the technical equivalence of Co-product Z and the displaced product(s)? • Review the substitution ratio(s) – are they in an acceptable range?
How would these products have been supplied?	Repeat method to assess how demand for displacing product is met	A separate analysis to establish the impact on yield, area, substitution and co-products for the affected product system(s) is required. This can be carried out using the same framework applied to biofuel feedstocks. The same rules on boundaries as described in section 3.3.1 apply.

3.3.4 Assessing the land use change and GHG consequences

3.3.4.1 GHG emissions associated with land use change

Exploring the impacts additional demand for a biofuel feedstock has on a range of product systems ultimately leads to the identification of land use changes that result in non-agricultural land being converted to agricultural production or in the avoided reversion of agricultural land back to natural land.

These land conversions entail loss (or in some cases gain) of carbon stocks which then needs to be converted to an estimate of the resulting GHG emissions. The amount of carbon stock change depends heavily on the geographical regions, the type of land converted and the management practices after conversion.

In determining the GHG emissions released as a result of land use changes, we used work carried out by Winrock International for the US EPA Renewable Fuel Standard 2010 or RFS 2 (US EPA, 2010).

As part of the calculation of ILUC impacts for the RFS 2, Winrock International used MODIS satellite data to estimate the different amount of land converted to cropland and pasture land in recent years in different world regions. The data provides estimates of proportions of different land types converted to cropland/pasture in different world regions, over a 6-year period (2001-7), based on satellite images taken at 500 m resolution. Winrock also carried out data validation by comparing satellite classifications with actual land types observed on the ground and through aircraft surveys and other satellite data. They also used NASA’s data validation information about which types of land MODIS tends to confuse, and used Monte Carlo analysis to correct for systematic misclassifications.

There have been extensive discussions as to the accuracy and appropriateness of this data for calculating ILUC impacts. Other global datasets could be used, e.g. FAO national statistics, although these datasets also have their drawbacks, e.g. forest data only being collected every 5-10 years, lack of consistency between countries and assessments, changing definitions of forest, different methods to avoid deforestation and unreliable and missing data (particularly for developing countries) (Olander et al., 2008).

However, as the main objective of this project was to focus on using the causal-descriptive approach to mapping market responses to increased demand for biofuel, rather than an in depth analysis of global trends in land use changes, the Winrock dataset was favoured for its global coverage and consistency. We are aware there are many very good datasets that could potentially be used to improve understanding of regional land use changes. For example, CBERS[4] for Brazil and China or the outputs from the TREES-3 Action on mapping changes in forest resources in Eurasian boreal forests and tropical forests[5].

Furthermore, the Winrock land conversion data provides estimates of the relative amounts of different types of land converted to cropland *and* pasture land combined. In this analysis, we specifically needed to know the amount of *cropland* that is expanding onto the different types of "natural" land. However, we felt that it was reasonable in most cases to use the Winrock conversion data for our purposes, for several reasons. First, in many cases cropland may expand onto pasture land as well as other cropped areas and cause pasture land displacement as well. We then used the simplifying assumption that the Winrock data for cropland *and* pasture land would capture this effect. Second, if obvious differences were observed between the evolution of a certain crop expansion onto cropland or pasture, we explored this further. For example, in the case of sugarcane ethanol (see chapter 8) we consider pasture intensification as a way of gaining land in Brazil and so we separate trends in sugarcane expansion onto cropland from trends in expansion onto pasture land.

Land use change thus results in large emissions (or in some instances uptake) of GHGs. While the change in land use happens over a short time frame, the changes in the carbon stocks and associated emissions / uptake can take up to several years, while agricultural products are grown on the land. All emissions / uptake over 30 years were added up to calculate the emission factors due to land use. These emissions factors were then annualised over 30 years again to calculate the final ILUC impact of biofuels in terms of GHG emissions per MJ of biofuel per year. In the case of palm oil biodiesel, a set of scenarios was explored in which emissions were annualised over a longer time period of 100 years. The point of these scenarios was to illustrate how land use change impacts are reduced, if the converted land is well managed and kept in productive use over the long term, instead of abandoned at the end of the plantation lifetime, as has happened for some palm plantations (Butler, 2006).

[4] http://www.cbers.inpe.br/?hl=en&content=introducao

[5] http://ies.jrc.ec.europa.eu/index.php?page=70

3.3.4.2 Indirect GHG emissions associated with biofuel induced yield improvements

This study does not include an "ILUC debit" for the GHG emissions associated with additional fertiliser used to achieve the above baseline yields in the biofuel projection, for the proportion of the crop that is not used for biofuel.

Yield improvements can be achieve through different agricultural practices, such as increased (nitrogen) fertiliser input, better management practices, use of new, higher-yielding varieties, etc. The GHG impacts of each of these practices are very different and depend on where these practices will take place and their uptake rate. If the above baseline yields by 2020 are realised largely through increased N fertiliser application, this could result in an overall increase in GHG emissions from the agricultural sector, as an increase in average yield would mean increased input of fertiliser for crops used for both biofuels and food. However, this is a particularly complicated issue, for the reasons described below.

Firstly, the relationship between N fertiliser and yield is not linear, and there may be a limit beyond which adding more fertiliser does not necessarily realise further improvements in yields, as shown in Figure 4.

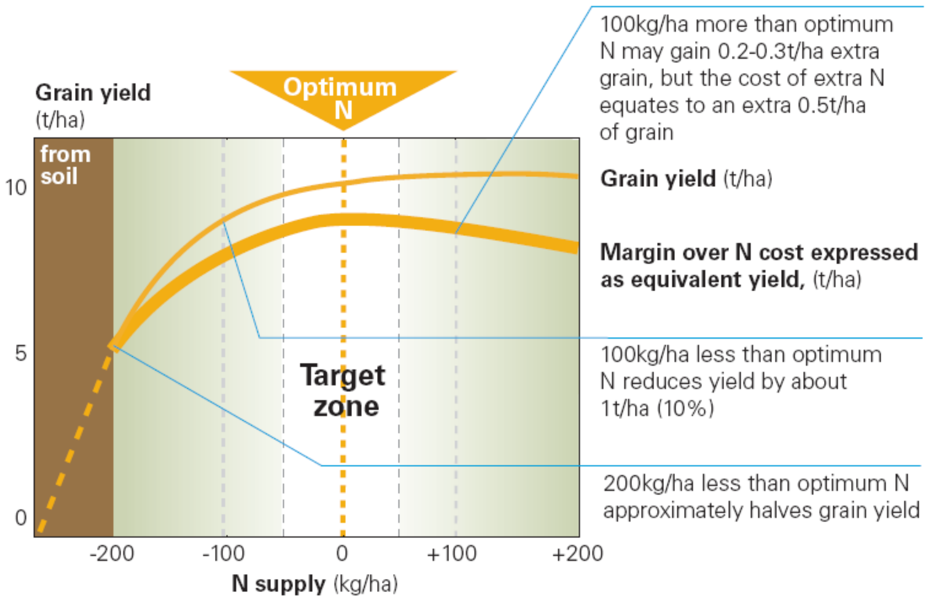

Figure 4. Wheat grain yield relative to N fertiliser supplied.

Source: HGCA, 2009.

However, how close a farmer is to that optimum depends on the region considered, the crop, the soil type, the climate, the farmer, etc. This is a level of detail which we have not been able to model in this project. As can be shown from the above graph, at low levels of N fertiliser applied, there is a more or less linear relationship between N supply and grain yield and therefore a small increase in N application does not result in an increase in kg N / tonne crop (wheat in the case of Figure 4). However, as more N is applied (the nearer it gets to the optimum supply of N), the linear relationship breaks down and the kg N applied per tonne of crop increases. It is this additional increase in N that is associated with additional GHG emissions; additional N supplied *per se* is not the issue but the supply of N/tonne crop.

Secondly, a higher protein content in the crop is not necessarily always the desired effect for farmers. For example, to make bioethanol from wheat, a higher starch content and lower grain protein in the wheat is more desirable. As shown in Figure 5, a lower grain protein and therefore higher starch content is associated with lower N supply. HGCA analysis (2007) found that the optimum N rate for alcohol production per ha was on average 12% lower than the economic optimum for grain production. Therefore, if less N fertiliser is required for wheat to be used for bioethanol (and the biofuel producer offers the farmer a premium for higher starch wheat), this could to some extent offset the additional fertiliser used in the food market to achieve the projected yield improvements.

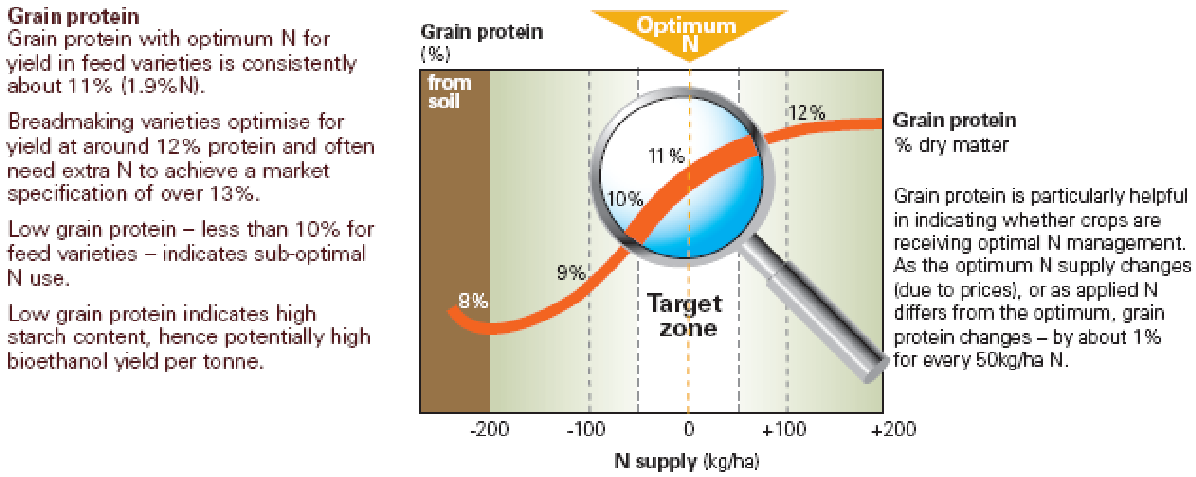

Figure 5. % Grain protein relative to N supplied.

Source: HGCA, 2009.

Finally, depending on the region of the world considered, different fertilisers are used, ranging from manure and urea to more industrial fertilisers such as ammonium nitrate. Different fertiliser types have different GHG emissions, both during their production process and during their application on the fields. For example, where chemical fertilisers are applied, urea has lower GHG emissions associated with its production (1.3 kg CO_2e / kg N) than ammonium nitrate (6.8 kg CO_2e / kg N), (Jenssen and Kongshaug, 2003). Therefore, projecting GHG emissions associated with increased fertiliser use not only requires knowledge of how much fertiliser is currently used, how much is likely to be used in the future and the yields associated with those levels of fertiliser use, but also the type of fertiliser that will be used in the regions where the crops are expanding.

For these reasons, it was not possible within the scope of this project to adequately model the additional indirect emissions associated with increased use of fertiliser. However, for illustrative purposes only, we have made a simple calculation of what the indirect GHG emissions associated with an increase in nitrogen fertiliser use for wheat would be if we made the following assumptions:

- average wheat yield of 7.76 t / ha (UK specific data taken from the RTFO– see RFA, 2009);
- an average fertiliser input of 183 kg N / ha (UK specific data taken from the RTFO see RFA, 2009);
- 10% increase in fertiliser input would lead to an increase in yield of 0.5 t / ha;
- The additional fertiliser applied was assumed to be ammonium nitrate with an associated emission factor of 6.8 kg CO_2e / kg N (RFA, 2009);

- The additional N_2O emissions from soil due to additional fertiliser input was calculated based on the IPCC Tier 1 methodology (De Klein et al., 2006).

Based on these assumptions, the additional fertiliser input results in additional GHG emissions of 8.9 g CO2e / MJ bioethanol. Approximately half (4.7 g CO_2e / MJ) are due to additional fertiliser production and distribution; the other half (4.2 g CO_2e / MJ) represent N_2O soil emissions. This value only takes account of indirect emissions – i.e. the emissions associated with additional (compared to the baseline) fertiliser input on fields producing food and feed products but not additional input on fields producing biofuels (as these should be accounted for under the direct emissions accounting scheme).

The GHG emissions associated with additional fertiliser input are not negligible. However, the emissions calculated here may well be at the higher end of the range of potential impacts, as yield improvements will also be achieved through better management practices, and the fertiliser assumed to be used is particularly carbon intensive. In the context of the overall ILUC factors calculated in chapters 4 to 8, this effect is may add somewhat to the indirect emissions, but is not likely to be so large to change the overall conclusions about the magnitude of the ILUC impacts.

3.3.5 Exploring uncertainty and alternative scenarios

Scenarios. One of the objectives of the project was to understand the sensitivity of the results to the main parameters and to management practices. Also, while conducting the analysis for all five chains, it became apparent that there are likely to be several possible outcomes in terms of ILUC impacts, depending on future contexts and system assumptions, and that it is difficult to characterise the ILUC impact using a single ILUC factor.

We have thus used scenario analysis to explore different possible likely outcomes. The different scenarios explored will be presented at the beginning of each of the chapters on specific chains.

Uncertainty ranges. Winrock International provides uncertainty ranges associated with the emission and reversion data developed for the US EPA (2010). To assess the importance of carbon stock data uncertainty on the calculation of ILUC factors, we have incorporated these uncertainty ranges in our analysis. For each of the biofuel chains, the results are presented as bar charts at the end of the relevant chapter. All bar charts show as error bars the uncertainty ranges due to uncertainty in carbon stock data.

3.3.6 Structure of the causal-descriptive model

The actual model was realised in Microsoft Excel. It is composed of several modules between which specific information and data flow. The main modules are:

- **Feedstock demand in 2020** – module in which the demand for feedstock in the baseline and in the biofuel projection in 2020 are defined.
- **Feedstock specific modules** – each of the five studied biofuel feedstocks has its own module, in which the market responses to an additional demand for the feedstock were modelled. Furthermore, the link between market responses and final GHG emissions due to land use change is also modelled in this module.
- **Co-product** – this module, based on data from the previous modules, examines the amount of additional co-product produced, the products displaced by these co-products and the

GHG credits associated with this displacement. The outcome of this module influences the demand for agricultural feedstocks in 2020 in the biofuel projection and thus feeds into the feedstock specific modules.

- **Land use data and emission factors** – this module is based on Winrock International data and allows for the conversion of market responses into land use change and GHG emissions in the feedstock specific modules.

Annex 9 presents a flow diagram presenting the main modules and the main data flows between these modules. The key point to highlight is that the model is iterative and does account for feedback loops: the production of biofuel from a certain feedstock, e.g. oilseed rape, will influence other agricultural commodities, e.g. wheat (through the displacement of wheat by rapeseed meal). This in turn will influence the demand for wheat in 2020, which changes the yield and area projections of this model. Another example is the influence of all feedstocks on the demand for palm oil, either through displacement of soybean oil by biofuel co-products or by substitution of vegetable oils out of the food market, which are assumed to lead to higher demand for palm oil in 2020. More background information on these assumptions is available in the following chapters.

3.4 Management practices for avoiding ILUC

Chapter 9 summarises management practices that we understand from our modelling would significantly reduce the ILUC impact of the different biofuel chains, if effectively implemented. Indicators are identified to present the outcomes that would illustrate that the management practice had been effective. Although not the key objective of this project itself, these practices and their effective implementation are important outcomes of this analysis as they represent a link between modelling these ILUC impacts and trying to mitigate them in a practical way. Chapter 9 also provides a useful link to other work that has been carried out by DfT and RFA on mitigation of ILUC effects.

4 Palm biodiesel

4.1 Introduction

This chapter discusses how we have modelled the ILUC impacts of global biodiesel production from palm oil in 2020 and how we have estimated the GHG consequences of those ILUC impacts.

The market responses associated with demand for palm oil for biodiesel are more straightforward to model than the other biofuel chains. This is because palm oil is considered to be the least cost vegetable oil and so increased demand for palm oil for biodiesel is most likely to result in increased palm production. Conversely, the soy and OSR chains both link to the palm chain, through palm oil substituting for those products in particular markets.

Scenarios. It is impossible to predict with any acceptable degree of certainty the exact ILUC impacts that will be associated with the demand for an additional MJ of palm biodiesel in 2020. However, it is possible to consider the factors that will be important in determining the ILUC impacts and how they could potentially vary in the future. We have therefore developed 10 different scenarios in which different permutations of the variation in these factors are considered. These scenarios are outlined in Table 8.

Table 8. Scenarios explored for ILUC impacts of palm oil biodiesel.

Scenario	1	2	3	4	5	6	7	8	9	10
16% yield increase attributed to palm biofuel demand	✓	X	X	X	X	X	X	X	X	X
No palm yield increase attributed to palm biofuel demand	X	✓	✓	✓	✓	✓	✓	✓	✓	✓
Historical deforestation	✓	✓	✓	✓	X	X	X	✓	✓	✓
10 % deforestation	X	X	X	X	✓	✓	✓	X	X	X
No peat land expanded onto	X	✓	X	X	✓	X	X	✓	X	X
5% peat land expanded onto	X	X	✓	X	X	✓	X	X	✓	X
33% peat land expanded onto	✓	X	X	✓	X	X	✓	X	X	✓
Single plantation lifetime	✓	✓	✓	✓	✓	✓	✓	X	X	X
Continuous plantings	X	X	X	X	X	X	X	✓	✓	✓

To summarise the key differences between the scenarios:

- Scenarios 2-4, consider historic levels of deforestation, a 30 year plantation lifetime, varying levels of peatland conversion and no additional yield improvement attributable to biofuel demand
- Scenarios 5-7 consider palm expansion onto only 10% deforested land, a 30 year plantation lifetime, varying levels of peatland conversion and no additional yield improvement attributable to biofuel demand

- Scenarios 8-10 consider historic levels of deforestation, continuous plantings over 100 years, varying levels of peatland conversion and no additional yield improvement attributable to biofuel demand
- Scenario 1 is the same as scenario 4, except with a 16% yield increase attributed to biofuel demand

This chapter starts with a description of how the additional demand for palm oil in 2020 was estimated. It then goes on to discuss in more detail in section 4.3 the different market responses to that increased demand. Section 4.4 discusses our approach for estimating the types of land that will be converted to cropland, the types of land that will no longer be converted to cropland and land that will revert from cropland to "natural" land, as a result of those market impacts. Section 4.4 also discusses the GHG impacts of those land use changes. Section 4.5 presents and discusses the results of this study. The final section outlines particular aspects of the analysis that could be studied in more detail by those developing further analysis of this kind.

4.2 Additional global demand for palm biodiesel in 2020

Section 3.2 outlines the approach used to project the amount of palm oil expected to be used globally in 2020 for both biofuel and other purposes. In order to understand the amount of palm oil for biodiesel that is *additional* to that used today, the current volume (2008) of palm oil used for biodiesel (taken from FAPRI, 2009) was subtracted from the projected volume of palm oil required for biodiesel in 2020.

An additional 16.6 million tonnes of palm oil are estimated to be needed globally in 2020 in the biofuel projection. This equates to 2.8-3.4m ha of land globally. For comparison, in 2008, 205.3 million tonnes of palm fresh fruit bunches (FFB) were produced globally, which equates to approximately 32.8 million tonnes of palm oil[6]. FAO estimates 14.6m ha of oil palm were harvested in 2008.

4.3 Market responses

The key producers and exporters of palm oil currently are Indonesia and Malaysia, and they are expected to continue being the main producers of palm oil in 2020. It was also considered by members of the Expert Advisory Group that South American countries such as Colombia would also be significant producers of palm oil in 2020. Assuming the average 4-year Compound Annual Growth Rate (CAGR) of palm production in Colombia over the period 1990-2008 (7.62%) continues out to 2020, Colombia will produce 2% of the additional palm oil required in 2020, and Indonesia and Malaysia combined would supply the remaining 98%. The split between the amount of palm oil supplied by Indonesia and Malaysia in 2020 was based on the FAPRI World Agricultural Outlook (2009) projection for 2018/19.

Table 9 summarises where it is assumed that additional palm oil will be produced in 2020, to meet the additional palm oil required for biodiesel.

[6] Assumes 16% oil extraction for 2008 compared with 20% OER in 2020.

Table 9. Location of additional palm oil planted in response to palm biofuel demand.

Country	% of demand
Indonesia	54%
Malaysia	44%
Colombia	2%

In each of the countries identified as marginal producers of palm oil in 2010, the additional palm oil required could be met in a number of different ways:

1. Increased area of palm plantations;
2. Increased yields of palm on existing plantations;
3. Substitution of palm oil in other markets (e.g. food, oleochemicals) by other vegetable oils;
4. Increased availability through improvements in efficiency in the supply chain;
5. Reduction in amount of palm oil demand in other markets.

The other market response to be considered is the effect that co-products from palm biodiesel production have on the markets in which they act as substituting products.

These market responses to the increased demand for palm oil for biodiesel are mapped out in Figure 6. The magnitude of the effect of the different market responses is discussed further in this section and illustrated through the scenarios.

As discussed in section 3.3.1, market responses 4 and 5 have not been considered in this analysis. The extent to which these market effects make a contribution could be explored in future analysis of this type.

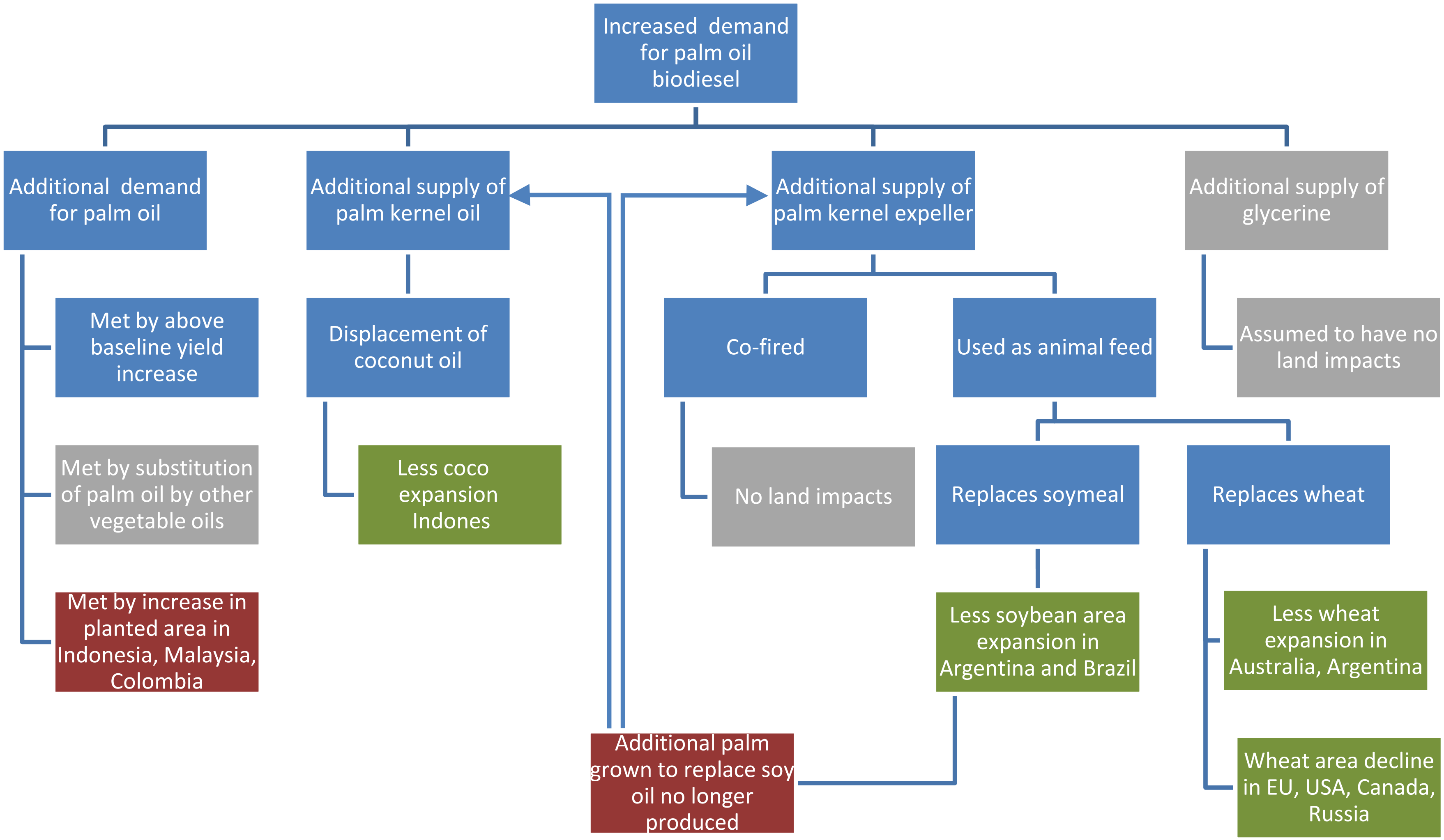

Figure 6. Market responses to an increase in demand for palm oil for biodiesel.

Land expansion, or ILUC "debits" are shaded red and avoided land expansion, or ILUC "credits" are shaded green. Market responses depicted in the diagram in grey are not considered further for reasons explained in the text.

4.3.1 Substitution of palm oil by other oils in non-biofuel markets

Vegetable oils (e.g. palm oil, rapeseed oil, soybean oil etc) are fungible products in many applications, i.e. they can substitute for one another. There was general agreement both in the literature (e.g. analysis by Schmidt and Weidema, 2008) and among stakeholders that palm oil is and is likely to continue to be the “marginal vegetable oil” in 2020. This is due to the lower price of palm oil compared with other vegetable oils. In other words, if there is an additional increase in the demand for vegetable oil, it is most likely to be provided by palm oil. So, if palm oil is diverted out of other markets and into the biofuel market, there is no reason to suppose that palm oil would be replaced by anything other than more palm oil. Therefore, in all palm oil scenarios, no product substitution effects are considered.

4.3.2 Biofuel demand induced yield improvements

In order to estimate the ILUC impacts of the wheat and OSR chains, we used the methodology developed in Lywood et al (2009a) and discussed in section 3.3.2.2 to estimate the effect that additional demand for these feedstocks for biofuel would have on increasing yields and expanding crop area. This methodology was also applied to palm oil. However, the results of applying this approach to palm oil were judged unlikely, as outlined below.

In summary, the results from using the Lywood methodology show that in the baseline, average yields would decline from 19.2 t FFB / ha in 2008 (averaged across Indonesia and Malaysia) to 17.2 t FFB / ha in 2020, compared with a decline to 18.7 t FFB / ha in the 2020 biofuel projection. The additional demand for palm for biofuel therefore results in an increase in 2020 of the yield of FFB/ha from 17.2 t FFB / ha to 18.7 t FFB / ha, or 16% of the increase in production. However, this outcome contrasts strongly with the consensus views from stakeholders that average yields of palm FFB are likely to increase in the future.

The predictive power of the yield/area modelling was tested using *ex ante* tests of root mean squared errors (RMSE) in Lywood et al. (2009a). Although their yield model seemed to predict better results than just assuming constant yield growth, based on a 15 year historic yield growth rate over the period 1977-1992, the predictive power of the yield model for SE Asia oil palm was not as strong as it was for EU cereals (RMSE of 1.36% for palm compared with 0.66% for EU cereals). This less good correlation may explain why this methodology predicts that yields will *decline* to 2020 in the baseline (i.e. without biofuels) *and* in the biofuel projection (although to a lesser degree).

In terms of understanding why the correlation between yield and area is not so good for oil palm, this may be explained by the fact that oil palm has only been grown commercially in Malaysia for example since the 1960s-70s, and that it has a long lifespan (25 years) compared with the other annual crops being modelled. As such, the dataset used to predict the relationship between yield and area is effectively based on one generation of plantings, and potentially the first generation of plantings, which is likely to make the outcome less robust.

Without any other similar mechanism for estimating the contribution of yields and area to meeting palm demand, it was assumed in most of the scenarios that the increase in demand for palm oil for biofuel would not induce any additional above baseline yield increases. The mechanism for estimating the baseline yield increase in palm is described in section 4.3.3.

However, we explored in scenario 1, the impact on an ILUC factor of the 2020 palm biofuel demand resulting in an additional 16% increase in yields above the baseline yield increase. However, this scenario is purely an illustration; it is not linked to the Lywood et al. (2009a) methodology, as it is not possible to combine our alternative approach for calculating 2020 yields with that methodology's approach for estimating the split between yield and area.

Alternative mechanisms for estimating the impact that additional demand for palm oil for biodiesel has on overall yields is clearly an important area for future study.

4.3.3 Yields used in calculating area expansion

As mentioned above, it was necessary to estimate the expected oil palm yield in 2020 via a different method than the Lywood methodology, as the yields projected for 2020 are much lower than the trends expected by experts.

The approach used was to estimate the average yield achieved in 2020 based on the additional area that would be planted with oil palm as a result of the demand for palm oil for biofuel. It was necessary to take into consideration the age profile in 2020 of all the oil palm planted in response to demand for palm oil biofuel, to understand what the average yield of that planted area would be in 2020.

The yield of FFB per hectare varies over the lifetime of oil palm plantation, as shown in Figure 7.

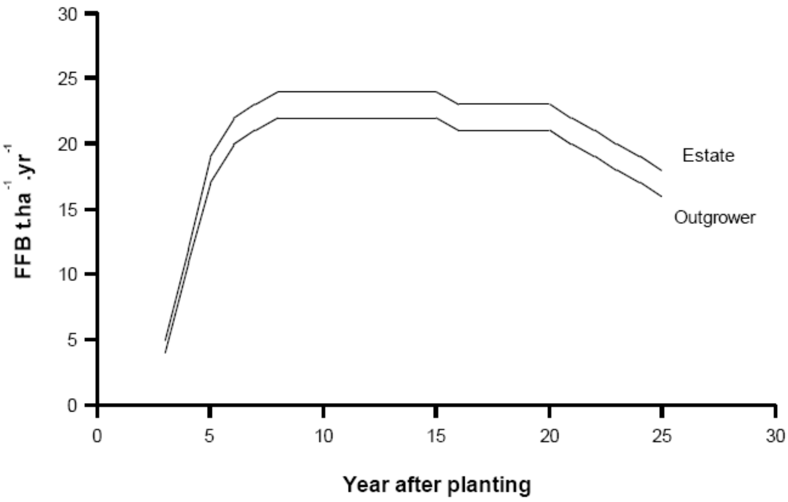

Figure 7. Palm yield profiles over the lifetime of a plantation managed by (a) and estate and (b) an outgrower.

Source: Chase and Henson (2010).

Based on feedback from stakeholders, it seemed reasonable to assume that the majority of expansion into new areas would be by large estate owners, rather than smallholders. It was also concluded that there would be no reason to assume that yields achieved on commercial plantations in Indonesia, Malaysia and Colombia would be significantly different and so these areas are considered together.

Jelsma (2009) showed that in 2006, the average yield achieved on the leading commercial plantations across Indonesia and Malaysia was 22.5 t FFB / ha. Assuming this yield was for a mature plantation at the peak of the yield curve in the above figure, the plantation would have been planted somewhere between 8-15 years previously (this is obviously a simplification, as there will be trees of different ages included in these plantations). Based on the above yield profile, the *lifetime average yield* of a tree planted 12 years previously in 1994, and achieving a maximum yield of 22.5t FFB / ha, would be 18.2 t FFB / ha.

We then assumed that the lifetime average yield of oil palm planted each year would increase by 1.5% per annum, based on historic rates of yield increase (Chin, 2004). Extrapolating to 2020 would mean lifetime average yields of palm planted in 2020 would be 26.8 t FFB / ha.

It was also assumed that initial additional oil palm planting in response to additional demand for palm for biofuel would have begun back in 2003, when there were the initial signals to the market that palm oil would be used for biodiesel production. We assumed a lower level of expansion for 2003-8 and a higher level of expansion between 2008 (when biofuel obligations started being introduced in Europe) and 2020. The rate of expansion was fixed between 2008 and 2020 and set to meet the production requirements for palm oil in 2020. As a result, each year, an extra area of palm would be planted and so in 2020, the additional planted area would have oil palm of different ages, and therefore different yields. The average yield in 2020 of all the different ages of oil palms planted indirectly in response to palm biofuel demand estimated using this approach is 22.9 t FFB / ha.

In terms of oil extraction from FFB (i.e. yield oil / t FFB), the extraction rate used in the RTFO is 16%. However, this was deemed too conservative for 2020, particularly as higher oil extraction rates (OER) of 21% are already being achieved (Wicke et al., 2008). As such, an OER of 20% was assumed.

4.3.4 Area increase

In the case in which a proportion of the palm oil is expected to come from yield increases induced by the additional demand for palm biofuel (scenario 1), the remaining palm oil required is assumed to be expanding onto new land. In the same way, in the scenarios in which none of the palm oil is expected to come from demand induced yield increases (scenarios 2-10), all the palm oil is assumed to be expanding onto new land. As discussed earlier, it is assumed that palm oil will be expanding in Indonesia, Malaysia and Colombia.

We have assumed that palm in the biofuel projection is unlikely to be expanding onto existing agricultural land in Indonesia and Colombia. In Indonesia, trends in land area harvested for other crops (according to FAO statistics, 2007) show there is no decline in other crops being grown; the area of oil palm is rapidly increasing, rubber increasing slightly, rice increasing, the rest more or less constant.

In Malaysia it may be the case that some palm will expand onto a small amount of cropland. According to FAO (FAO statistics) the Malaysian oil palm area is rapidly increasing, and the area planted with other main crops declining slightly (rubber) or constant (rice). Palm has historically expanded onto some rubber plantations in Malaysia that have been abandoned. However, we have assumed that this proportion of land is relatively small and that if it is used, it will be used in the baseline projection for other crops. The assumption is therefore that the expansion onto old rubber plantations is not something that will specifically result from increasing demand for palm oil for biofuel. In other words, we assume that all additional palm in the biofuel projection in Malaysia expands onto non-agricultural land. This assumption does not affect the results greatly. If we had assumed that palm expanded onto old rubber in the biofuel projection, we would have to make an assumption about what would happen to that land in the baseline. Either that land would revert to natural land or it would be used for other crops. For the former, we would use an avoided deforestation (reversion) factor to calculate the emissions associated with the LUC. For the latter, we would have to assume that those crops would be displaced to somewhere else (ultimately new land) and would also use a conversion factor to calculate the emissions associated with the LUC. Although there are differences in the emissions factors for conversion to annual crops, conversion to perennial crops and avoided reversion, they would not make a significant difference, particularly given the relatively small area of land being considered.

4.3.5 Effect of co-products

In all the biofuel production chains considered, there are other products that are co-produced. For example, in the palm oil biofuel production chain, the three main co-products that are considered are palm kernel oil, palm kernel expeller and glycerine. There are other co-products that are produced, e.g. empty fruit bunches, but these are considered to be consumed in normal practice within the boundaries of the system, e.g. returned to the soil for fertiliser or burned for power at the processing site.

In the cause/effect (or consequential) type of analysis being considered in this study, it is important to consider the land and related GHG impacts of the additional production of these co-products. For example, Table 11 summarises what is likely to happen to the products that are co-produced with palm oil biodiesel.

However, a number of different biofuel chains will result in co-products that displace the same product. For example, palm kernel meal, rapeseed meal and wheat DDGS all substitute soybean meal as an animal feed to a greater or lesser extent. Given the ratios of displacement assumed for soybean meal by different biofuel co-products (see Table 10), it is possible to calculate the reduction in soybean that needs to be produced as a result.

Table 10. Ratio of co-product displacement of animal feeds.
Displacement ratios taken from Lywood et al. (2009b).

Co-product	t co-product/t feedstock	Displacement ratio of soybean meal (t soybean meal/t co-product)	Displacement ratio of feed wheat (t wheat/t co-product)
Rapeseed meal	0.57	0.605	0.145
Palm kernel expeller	0.03	0.128	0.606
Wheat DDGS	0.33	0.594	0.386
Corn DDGS	0.31	0.395	0.494

However, calculating the area of soybean or wheat that no longer needs to be grown is slightly more complex. Using the Lywood et al. (2009a) methodology, (described in more detail in section 3.3.2.2), the displacement by co-products means less demand for the crops displaced, which has an impact on demand induced yields and area expansion. The impact on area therefore needs to be calculated for the biofuel projection considering the effects of all co-products from all biofuels.

The geographical location in which the soybean or wheat will no longer be grown also needs to be identified. Soybean is mainly produced in South America, particularly in Argentina and Brazil, therefore the avoided soybean cultivation was assumed to be equally split between these two countries. For wheat, we looked at where the co-product was produced. If it was produced in Europe, wheat cultivation in Europe would be avoided. If the co-product was produced outside of Europe, wheat cultivation was assumed to be avoided in the five main wheat exporting countries excluding the EU (i.e. the US, Russia, Australia, Canada and Argentina) based on the share of wheat exports of each of these countries.

It is important to have an accurate baseline for including land savings from co-products. For example, if a co-product is credited with avoided deforestation, it would be important to make sure that the effectiveness of external measures aimed at avoiding deforestation, such as through REDD, are included in the baseline. This is important for ensuring that the ILUC factor only credits avoided deforestation that is actually happening.

Table 11 summarises the assumptions made about the products that the palm biofuel co-products substitute.

Table 11. Market displacement effects of products co-produced with palm oil biodiesel.

	PKO	PKE	Glycerine
Market	Global	• 58% EU market • 42% rest of world	Global
Displaces	Coconut oil	• Animal feed (soy, wheat) • Coal (can be co-fired)	Many uses – but not assumed to displace land based products in this analysis
Ratio of displacement	1t coconut oil / t PKO	• Animal feed: o Soy: 0.13t soy/t PKE o Wheat: 0.6t wheat/t PKE • Coal (not considered as no ILUC impacts)	
Country of displacement	Indonesia	• Soy: Argentina, Brazil • Wheat: EU (for EU consumption), US, Russia, Australia, Canada and Argentina (for other consumption) • Coal: Europe	

4.3.5.1 Palm Kernel Oil (PKO)

It is assumed in this analysis that the additional production of PKO would displace coconut oil. The view from our expert advisory group was that although in the countries where it is produced, PKO and palm oil can be used interchangeably, the very different properties of lauric oils, like PKO, compared to palm oil, mean that in other markets there are practically no instances of substitution. Coconut oil and PKO are both known as lauric oils because they are high in lauric acids, which give them very similar physical and chemical characteristics. They are both used in the same applications (e.g. oleochemicals and soap, specialty foods, cooking oils in Indonesia, Malaysia and Philippines) and are generally considered to be interchangeable.

Indonesia and the Philippines are currently by far the largest exporters of coconut oil and Indonesia is also a large exporter of palm oil (FAO export statistics, 2007). The view from our expert advisory group was that these countries would remain the dominant exporters in 2020. The view was that if PKO expands as a result of palm oil production, Indonesia would be most likely face price competition for its coconut oil from its own PKO exports, thereby making them the marginal producers of coconut oil. It was also argued that these coconut oil producers would be able to respond quickly to market price signals for coconut oil. The majority of coconut production in Indonesia is on small-holdings and small holders can react quickly to falling price pressure by felling their trees in times of low coconut pricing, sell the timber and then re-plant. Whilst the new trees are becoming established, the smallholding is inter-planted with alternative cash and/or subsistence crops. Conversely, in times of increasing demand for coconut oil (i.e. in the baseline, without biofuels), the assumption would therefore be that new coconut plantations would be need to be planted, presumably inter-planted with other crops while the plantation is being established. The biofuel projection in this case would need to be credited with the avoided expansion of coconut plantation in the marginal producing country of Indonesia.

However, the palm ILUC factor calculated is clearly very sensitive to this assumption. If Brazil was assumed to be the marginal producer of coconut oil, a much larger proportion of grassland might be expanded onto than is observed in Indonesia. Also, because coconut is a perennial tree crop, expansion onto grassland could result in an increase in the carbon stock of the land. Thus, displacing this production might, in those specific instances, result in a carbon debit instead of a carbon credit. Therefore, although the most likely marginal producer has been chosen in this analysis, further analysis could be done in the future to explore alternative scenarios.

4.3.5.2 Palm Kernel Expeller (PKE)

It is estimated that 58% of PKE produced would be imported into the EU market and 42% imported by the rest of the world (based on current trends in PKE imports as per FAO 2007 import statistics). It was important to make this distinction because it is assumed that in the EU, 50% of the imported PKE would be used for co-firing with coal in power stations and 50% in the animal feed market, whereas outside the EU, it was assumed that all PKE imported would be used in the animal feed market. The PKE that is co-fired is not assumed to have any ILUC impacts associated with it, and is therefore not considered further.

The PKE used for animal feed is assumed to displace soybean meal in a ratio of 0.13 t soybean meal / t PKE and feed wheat in a ratio of 0.6 t wheat / t PKE. These ratios are taken from Lywood et al. (2009b) and are calculated so that the digestible energy and digestible protein values of soy and cereal feed components removed are equivalent to those in the co-products added in the feed.

There is a knock-on effect of PKE replacing soybean meal however. This is that soybean meal is the determining co-product of soybeans and that soy oil is the dependent co-product. In other words, soybeans are grown for the meal rather than the oil (see section 6.3.1 for more discussion of this issue). Therefore, if soybeans are no longer produced, there will be a reduction in the amount of soy oil produced. If this soy oil is no longer produced, it is assumed in this case, for simplicity that it will need to be replaced by the least cost vegetable oil, which is palm oil. Therefore more palm will need to be planted (although this is not a significant amount due to the relatively small proportion of soybean meal that PKE substitutes), and there will therefore be more PKO and PKE co-products. The PKE will then displace more soybean meal and reduce further the amount of soy oil produced, etc. However, this effect does not result in an unending loop but can be solved as an equation with two unknown parameters, as shown by Weidema (1999) among others. The additional impact of this effect is actually very small and does not result in a significant reduction in the amount of soy meal grown or significant increase in the palm expansion.

4.3.5.3 Glycerine

As for the other biofuel chains, the land impacts of the glycerine co-product produced in the palm biofuel chain are not taken into consideration. This is because although glycerine has many uses (e.g.

pharmaceutical and cosmetics) these are not assumed to displace products that would be grown on land. There have been discussions recently about the potential for using glycerine as an animal feed and if this is the case, this could potentially result in land savings. This could be investigated further in future analysis.

4.4 Land use impacts and greenhouse gas consequences

4.4.1 Types of land converted

4.4.1.1 Palm expansion

In Table 12 the types of land being converted to cropland/pasture land in the countries affected by increased demand for palm biodiesel are shown. These land conversions are based on the information provided by Winrock in the EPA Regulatory Impact Assessment (US EPA, 2010). In the countries considered below, all palm expansion is expected to be on previously non-arable land (see section 4.3.4).

There was some discussion in the stakeholder meetings and in the feedback from interested parties, as to whether it was appropriate to use historical trends in land conversion in these countries because they are likely to have better forest protection in 2020 (through commitment to programmes such as REDD etc) and that historic deforestation rates are unlikely to continue. One set of scenarios (scenarios 5, 6 and 7) therefore model a situation in which much better forest protection could be achieved. In these scenarios it was assumed that in Malaysia and Indonesia, deforestation could be reduced to 10% of land converted to cropland, to illustrate the sensitivity of the ILUC impacts to assumptions about deforestation.

In Indonesia it is assumed in these scenarios that the palm no longer produced on forest land would instead be produced on grassland. In our model, this would equate to an expansion onto 0.76m ha of grassland. According to IIASA (2009), Indonesia has 8m ha of imperata grassland and 3m ha of unprotected grassland and woodland that is very suitable for oil palm.

In Malaysia, there is not the large area of imperata grassland that there is in Indonesia. However, there are 10m ha of land categorised as “mixed and cropland” according to WRI (2003). FAO record 6.5 million ha of cropland in Malaysia in 2008, suggesting 3.5 million ha of the remaining land could be categorised under the “mixed” land category. It is therefore assumed that palm no longer produced on forest land (in these scenarios) in Malaysia would instead be produced on mixed land. In our model, this would equate to an expansion onto 1.1 million ha of mixed land.

Table 12. Types of land palm is expands onto in the marginal producing countries in 2020.
Based on Winrock data for the US EPA (2010).
[Numbers in parentheses indicate the scenario in which less forest is assumed to be converted and more other type of land converted instead].

Type of land converted to crop/pasture:	Forest	Grass	Mixed	Savannah	Shrub land	Wetland	Barren
Indonesia *[good forest protection scenario]*	39% *[10%]*	5% *[34%]*	29% *[29%]*	22% *[22%]*	3% *[3%]*	2% *[2%]*	0% *[0%]*
Malaysia *[good forest protection scenario]*	52% [10%]	3% [3%]	27% [69%]	13% [13%]	2% [2%]	2% [2%]	0% [0%]
Colombia	33%	9%	31%	18%	8%	1%	1%

4.4.1.2 Land impacts of co-products

The same Winrock emissions factor dataset was used to explore the types of land that are no longer expanded onto as a result of the palm co-products displacing the expansion of certain crops (see Table 13). Also, Winrock's *reversion* dataset (the amount of cropland reverting to other different categories of land) was used for those regions where the palm co-products resulted in a *decline in arable crop area* through substitution of the crops, rather than an *avoided expansion* of the crops (see Table 14).

For displacement of feed wheat by PKE, the emissions factor dataset or the reversion dataset was used depending on whether avoided expansion was taking place or reversion in a particular region.

- In Europe, USA, Canada and Russia, the total wheat area was assumed to decline to 2020 in our biofuel projection. The assumption was then made that the reduction in wheat area would translate into a reduction in arable area, which would then revert to "natural land".
- In Argentina, Brazil and Australia, arable area was projected to increase and so emissions factors were used instead of reversion factors.

The assumption that total wheat area declining is an indicator of reduction in arable area is a simplification and is an area for further analysis in the future.

Table 13. Relative proportions of types of land no longer expanded onto as a result of palm biofuel co-products.
Based on Winrock data for the US EPA (2010).

Avoided expansion of cropland/ pasture onto:	Reason	Forest	Grass	Mixed	Savannah	Shrub land	Wetland	Barren
Indonesia *[good forest protection scenario]*	Avoided coconut expansion	39% *[10%]*	5% *[34%]*	29% *[29%]*	22% *[22%]*	3% *[3%]*	2% *[2%]*	0% *[0%]*
Argentina	Avoided soybean or wheat expansion	12%	26%	27%	17%	14%	1%	3%
Brazil	Avoided soybean expansion	55%	9%	16%	8%	7%	9%	0%
Australia	Avoided expansion of wheat	6%	32%	11%	22%	25%	0%	4%

Table 14. Relative proportions of types of land that cropland will revert to, as a result of declining cropland areas due to displacement of crops with biofuel co-products.
Based on Winrock data for the US EPA (2010).

Reversion of cropland/ pasture to:	Reason	Forest	Grass	Mixed	Savannah	Shrub land
EU	Reduction in wheat area grown	45%	11%	15%	17%	11%
United States	Reduction in wheat area grown	27%	31%	14%	12%	16%
Canada	Reduction in wheat area grown	43%	11%	4%	15%	26%
Russia	Reduction in wheat area grown	43%	10%	6%	14%	27%

4.4.2 Emissions from land converted

4.4.2.1 Winrock datasets

Alongside the Winrock satellite dataset (US EPA, 2010), a database of emissions factors for types of land converted was published. This dataset provided, with 95% confidence intervals, a quantification of the GHG emissions (or uptake) resulting from the conversion of different land types to either annual or perennial cropland (or the reversion of cropland to different land types).

The Winrock emissions factors for converted land take into consideration changes in above and below ground carbon stocks, soil carbon, foregone sequestration (it assumes forests continue to accumulate carbon over their lifetime), and CO_2 and non-CO_2 emissions from burning converted land (only in regions where this is assumed to continue to take place).

4.4.2.2 Emissions from expansion onto peatland

The Winrock emissions factors take into consideration emissions from converting peatland through assuming a proportion of land in each sub-region of certain countries (e.g. Indonesia and Malaysia) is peatland. However, emissions from peat are only included for conversion of peatland to annual crops and not to perennial crops.

As conversion of peatland to perennial crops also requires peat drainage, scenarios (numbers 1, 3, 4, 6, 7, 9, 10) were developed in which the Winrock conversion factors were adapted to include emissions from conversion of peatland to perennial crops (i.e. palm).

The emissions included for conversion of peatland to perennial crops were based on analysis provided by the JRC, and based on the following assumptions:

- That it is appropriate to use subsidence measurements as a basis for estimating the average rate of peat oxidation
- The initial depth of peatland drainage is 80 cm (as per the Indonesian decree which specifies that drainage should be limited to this depth for "sustainable" production of palm oil)
- That for drainage depth over 50 cm, subsidence tends to level off at about 4.5 cm/yr (based on Couwenberg, 2009)
- That peat oxidation is responsible for 60% of the subsidence measurement (with compaction responsible for the remainder of the subsidence) (Wösten, 1997)
- A bulk volumetric carbon density of peat of 0.068 g C/cm^3, (as per Couwenberg, 2009)
- A symmetrical uncertainty range

Given these assumptions, the rate of CO_2 loss from peat oxidation based on subsidence measurements is estimated at 57 +/- 12 t CO_2 / ha.yr[7].

As mentioned above, the Winrock dataset did assume a proportion of expansion onto peatland for annual crops, which could be used as a proxy for expansion of palm onto peatland in those regions. The average proportion of peat assumed to be expanded onto for annual crops for both Malaysia and Indonesia was ~5%.

However, evidence brought forward by stakeholders suggested that the proportion of peatland that could be expanded onto could be significantly higher:

- The "Tropical Peat Research Institute" (TPRI 2009) (quoted in "Status of Peatlands in Malaysia" July 2009 report by Wetland International), displayed a conference poster showing that that the area of oil palm on peatlands in Sarawak increased by roughly 200,000 ha between 2003 and 2008. Over the same period, palm expanded onto 640,000 ha across Malaysia, according to FAO production statistics. Using these figures, the proportion of oil palm planted on peatland could be 30% (assuming no expansion onto peatland in other regions of Malaysia).
- Hooijer (2006) estimates that in Indonesia in 2006, 25% of timber and oil palm concessions were on peatland

Three sets of scenarios were therefore considered:

1. No peatland was expanded onto

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[7] Personal communication with Robert Edwards, JRC (2010).

2. 5% of land expanded onto in Indonesia and Malaysia is peatland
3. 33% of land expanded onto in Indonesia and Malaysia is peatland

Using these different scenarios helps provide an understanding around the sensitivity of the ILUC impacts to the extent of expansion onto peatland.

4.4.2.3 Timescales

30 years is the timeframe over which ILUC emissions from growing palm oil for biodiesel are amortised in the majority of these scenarios. For palm this is a suitable timeframe because it covers one lifetime of a palm plantation (usually estimated to be ~25 years). The profile of emissions from converted land is usually that there is a large peak of emissions in the first year, associated with the removal of carbon stocks, e.g. the timber, followed by a tail of emissions for the following decades, which are associated with emissions from soil and from subsidence of peatland (if grown on peatland). Foregone carbon accumulation of forests is also taken into consideration over the timeframe considered.

However, fundamental to the carbon balance is also how the carbon stock associated with the palm plantation is accounted for. Two sets of scenarios are considered which take into consideration different assumptions about timeframes.

(a) 30 year time frame – this assumes only one palm plantation lifetime at the end of which the plantation is removed. Therefore, at the end of the 30 year period, only a small amount of carbon would be left on the land as per the Winrock emissions factors calculations (equivalent to 15 +/- 75 t CO_2 / ha), as the majority has been removed at the end of the useful lifetime of the plantation

(b) 100 year time frame – this assumes palm would be continuously grown (successive plantings) on the land that is expanded onto. The average carbon stock on the land would be much higher in this case to reflect the fact that oil palm trees would be present on the land over the 100 year period. The assumed carbon stock on the land after conversion is equivalent to 129.3 +/- 40.3 t CO_2 / ha (Germer and Sauerborn, 2008), which is also similar to the estimated mean *in situ* standing plantation C stock of a plantation studied by Henson (2009).

For both scenarios it is assumed that any carbon in timber products would be converted to CO_2 over the 30 or 100 year timeframe. More discussion of this issue can be found in US EPA (2009), section 2.4.4.2.6.5.

4.4.2.4 Emissions attributable to logging

It is important to make the distinction as to whether any emissions from forest conversion will be attributed to the logging activity that takes place to remove high value timber from the forest before clearing for planting palm. If emissions from forest clearing are attributed to the timber products from logging, a small proportion of the emissions will be attributed to the palm oil. IPCC estimates the carbon stock of a pristine forest in insular Asia to be of the order 275 t C / ha. However, to account for the selective logging, it is more appropriate to use the standing biomass of a partially logged forest. The carbon stocks of forest in the Winrock emission factors are considerably lower than this for Indonesia and Malaysia (averaging at 143 t C / ha and 130 t C / ha respectively for those forested regions in those countries). The dataset used by Winrock (Brown, 2001) has used GIS data to estimate carbon densities of forests. Given the difference between the carbon stock in a pristine forest and those estimated by Brown (2001), it is reasonable to assume that the dataset used by Winrock already corrects for selective logging. As such, it is assumed that the emissions attributable to selective logging are not attributed to palm oil in this analysis. However, the emissions associated with the removal of the remaining timber in the forest *are* attributed to the palm chain.

4.4.2.5 Emissions and reversion factors

Emissions factors were therefore calculated for 30 years and 100 years for the different types of land conversion and these are shown in Annex 2. It should be noted that Winrock calculate these emission and reversion factors by sub-country region, so it would be possible to only use the emission and reversion factors for the country regions most likely to grow a certain type of feedstock. This analysis has not been undertaken here and averages are provided for each country.

4.5 Scenario results

The following graph shows the different ILUC factors estimated using our methodology for the different scenarios explored.

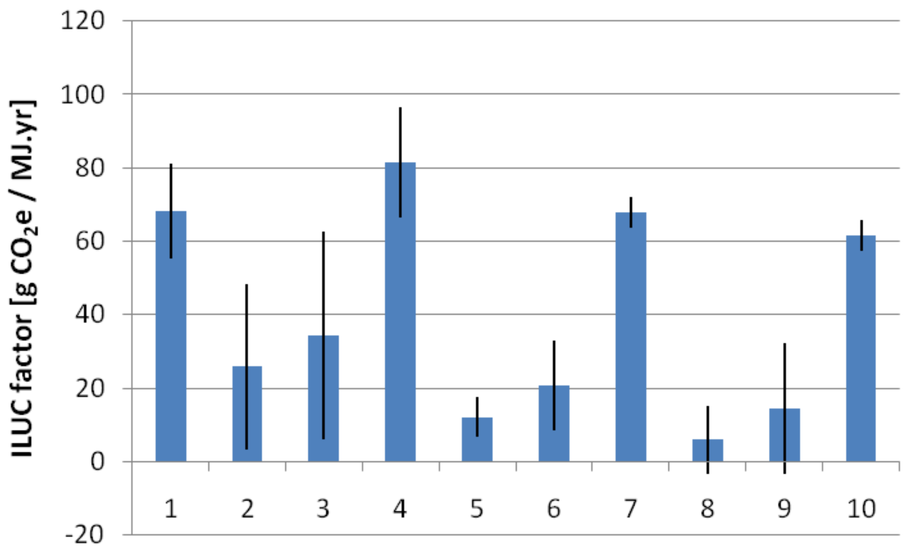

Figure 8. Indirect land use change impacts for the different scenarios modelled for palm biodiesel.

As can be seen from the graph, there are a wide range of potential ILUC impacts for the palm chain, depending on the assumptions made. To summarise these effects:

- The amount of peatland assumed to be expanded onto has the biggest impact on the ILUC factor (note the large ILUC impacts in scenarios 4, 7 and 10, where 33% of land expanded onto is assumed to be peat land)
- Whether palm expands onto forest or onto land with lower carbon stocks has a very important effect (contrast scenarios 2-4 with 5-7)
- Whether palm is grown continually on an area of land over the long term (e.g. 100 years) or if it will be abandoned after 1 planting also has a very important effect (contrast scenarios 2-4 with 8-10)
- The contribution of biofuel demand induced yields to the additional production of palm oil in 2020 can have a significant effect (contrast scenarios 1 and 4).

The uncertainty bars on the graphs represent estimates of the uncertainty associated with the carbon stock changes on the land and are calculated from Winrock (US EPA, 2010)[8]. The magnitude of the uncertainty, coupled with the number of possible scenarios illustrates the challenge associated with estimating with any precision the ILUC impacts of using a MJ of palm oil biodiesel.

Figure 9 illustrates the contribution of the different market responses to the final ILUC factor. Scenario 3 is shown as an example (a breakdown of the ILUC factors for the other scenarios are included in Annex 3). It represents the case in which historical rates of forest conversion continue, only 5% of peatland is expanded onto in Indonesia and Malaysia, the palm is only assumed to be grown on the land for one planting (and therefore the emissions are amortised over 30 years) and palm biofuel does not induce above baseline yield increases. The ILUC debits (i.e. carbon stock losses) are shown in red and the ILUC credits (i.e. avoided carbon stock losses through use of co-products) are shown in green. The final ILUC factor for this scenario is shown in black.

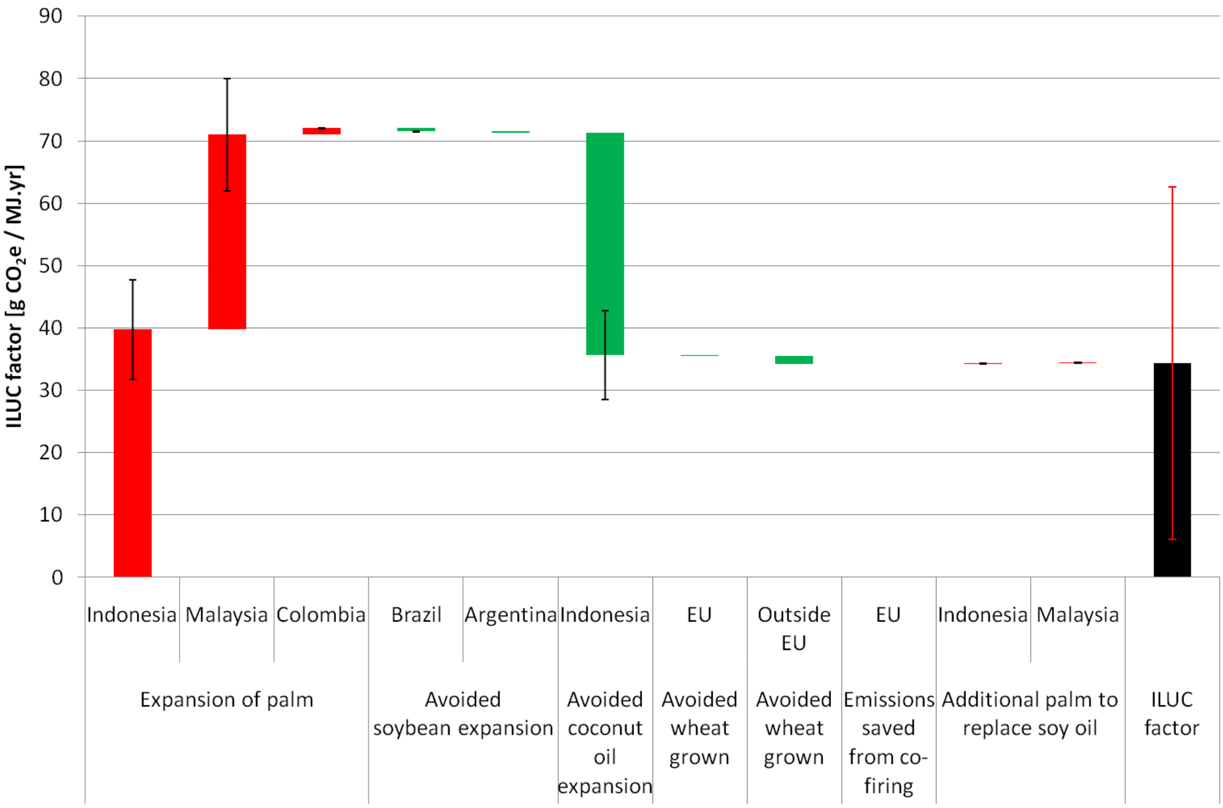

Figure 9. Waterfall diagram showing the contribution of each market to the overall ILUC factor for scenario 3.

Important factors to note from this waterfall diagram include:

- The credit attributed to PKO displacing coconut oil production in Indonesia. PKO is assumed to be produced in a ratio of 0.025 t PKO / t FFB and PKO is assumed to replace coconut oil in a ratio of 1:1. With approximately 2 m tonnes of PKO produced in this analysis, and a coconut oil yield of

[8] Apart from the data points for emissions from peatland and the carbon stock on the land for successive plantings of oil palm (see section 4.4.2).

1.18 t oil / ha[9] this equates to approximately 1.7 m ha of land saved in Indonesia, which is similar to the area of land expanded for palm oil in Indonesia – hence the similar size of the credit attributed to the PKO. This credit is large in this scenario because it was assumed that a large proportion of the land that would be converted to coconut oil production in Indonesia was forest land (the same proportion that is assumed to be converted to palm oil)

- The credits associated with displacing soy and wheat production are relatively small because of the lower substitution ratios (compared with the 1:1 substitution ratio between PKO and coconut oil), the lower proportion of forest land that is converted in Argentina and Brazil compared with that assumed for Indonesia and Malaysia and also the lower emissions factors associated with conversion of Argentinean forest and Brazilian forest relative to Indonesia and Malaysia.

From the different scenarios explored, it is clear that certain practices in the palm industry would reduce the ILUC impacts associated with using palm oil (for all products, not just biofuels):

- As illustrated in scenario 1, improved yields can have the benefit of resulting in less area expansion. However, it is important to ensure that these improved yields are achieved through means other than increased nitrogen application. Indications from stakeholder workshops were that there were indeed options to improve oil palm yields without energy and carbon intensive inputs.
- Expansion onto peatland dramatically increases the CO_2 emissions.
- Expansion onto forested land also has a large impact on the emissions from land use change. Expansion onto grasslands (that are not high in biodiversity) or other lower carbon stock lands could be one option for reducing the impacts associated with land use change for oil palm production.
- Ensuring land is sustainably converted to palm plantations for the long term and that the land will not be abandoned after one planting – i.e. ensuring a long term C stock is established on the land.
- Ensuring that co-products are used effectively to replace other land based products.

[9] This is derived from historical trends in CAGR for coconuts in Indonesia from 1961-2008 (1%) to give a coconut yield in 2020 of 7.1t coconut/ha. A copra yield of 0.249t copra/t coconut and an oil yield 0.67t coconut oil/t copra were assumed.

5 Oilseed rape biodiesel

5.1 Introduction

Oilseed rape is grown to produce rapeseed oil, which is usually used in the food market, but can also be transformed into biodiesel through an esterification process. It is currently the most important feedstock for biodiesel production in Europe and is likely to remain so in the near future. Today, most of the European demand for oilseed rape is produced domestically, mainly in France, Germany, Poland, the Czech Republic and the United Kingdom (FAOSTAT, 2010a).

In the following sections, we will discuss the indirect land use change impact of biodiesel production from oilseed rape in 2020 and its GHG consequences.

Oilseed rape has a number of peculiarities that need to be considered in determining ILUC impacts. It is a break crop, and not considered to be economically viable to be grown on its own. It is therefore included in crop rotations with cereals, usually wheat. This has important implications as an increase in oilseed rape harvested area is unlikely to mean expansion of oilseed rape cultivation area onto other types of land. Increase in harvested area of oilseed rape has to be assessed in combination with the cereal with which it is grown in rotation.

The oilseed rape chain is intimately linked to other vegetable oil chains, particularly to palm, through the possible replacement of rapeseed oil by palm oil in food markets, and to soy, through the use of rapeseed meal as replacement of soybean meal.

Scenarios. To reflect the many possible pathways through which oilseed rape biodiesel can lead to ILUC, several scenarios have been considered when estimating the impact on land and related GHG consequences. Table 15 below shows how parameters have been varied between the different scenarios. The exact numerical changes to the model for each scenario are discussed in the following sections.

Table 15. Overview of scenarios for the oilseed rape to biodiesel chain.

Scenario / Parameter	1	2	3	4	5	6
Amount of oilseed rape produced in Europe	High	High	High	High	High	Low
OSR displaced out of Ukrainian food market	No	Yes (50%)	No	No	No	No
Deforestation rates in Indonesia and Malaysia	Historical	Historical	Historical	10%	Historical	Historical
Share of rapeseed meal used as animal fodder	100%	100%	50%	100%	100%	100%
Sources of co-product substitution ratios	Lywood et al. (2009b)	Lywood et al. (2009b)	Lywood et al. (2009b)	Lywood et al. (2009b)	JEC (2008)	Lywood et al. (2009b)

This chapter starts with a short description of the baseline and biofuel projection for oilseed rape. The possible market responses to the additional demand for oilseed rape biodiesel are then discussed in section 5.3. Section 5.4 analyses the land use impact of the market responses and their GHG consequences, and section 5.5 presents the results of the analysis.

5.2 Additional demand for oilseed rape biodiesel in Europe in 2020

Based on data from FAPRI (2009), the demand for oilseed rape in 2008 in Europe (including the amount of oilseed rape used for biodiesel production[10]) was 19.1 million tonnes (see section 3.2).

Our baseline for oilseed rape was built assuming no additional demand for oilseed rape in Europe will occur between 2008 and 2020. We felt this was a sensible assumption given the slight increases or decreases that have been used in other "no biofuel" projections. Thus the baseline demand for oilseed rape in 2020 is also 19.1 million tonnes.

As shown in section 3.2, the additional demand for biodiesel in 2020 will be 23 billion litres. Oilseed rape is estimated to contribute 41% of the biodiesel demand in 2020, corresponding to a demand for oilseed rape of 26 million tonnes in 2020.

5.3 Market responses

In Europe, the additional demand for oilseed rape could be met in a number of different ways:

1. Increased supply of oilseed rape in Europe, either through substitution of rapeseed oil in the food market by other vegetable oils or by increased production of oilseed rape in Europe;
2. Changes to the European trade balance of oilseed rape, either through increased imports or lower exports;
3. Increased availability through improvements in efficiency in the supply chain;
4. Reduction in amount of rapeseed oil demand in other markets.

The other market response to be considered is the effect that co-products from oilseed rape biodiesel production have on the markets in which they act as substituting products.

These market responses to the increased demand for oilseed rape for biodiesel are mapped out in Figure 10. The magnitude of the effect of the different market responses is discussed further in this section and illustrated through the scenarios.

As discussed earlier in this report, market responses 3 and 4 have not been considered in this analysis. The extent to which these market effects make a contribution could be explored in future analysis of this type.

[10] The share of oilseed rape grown for biodiesel production in Europe was 64% in 2008 based on FAPRI (2009) statistics. These are consistent with other sources such as USDA FAS (2010) (66% if industrial end-use is taken as a proxy for biodiesel production) and Fischer-Boel (2008).

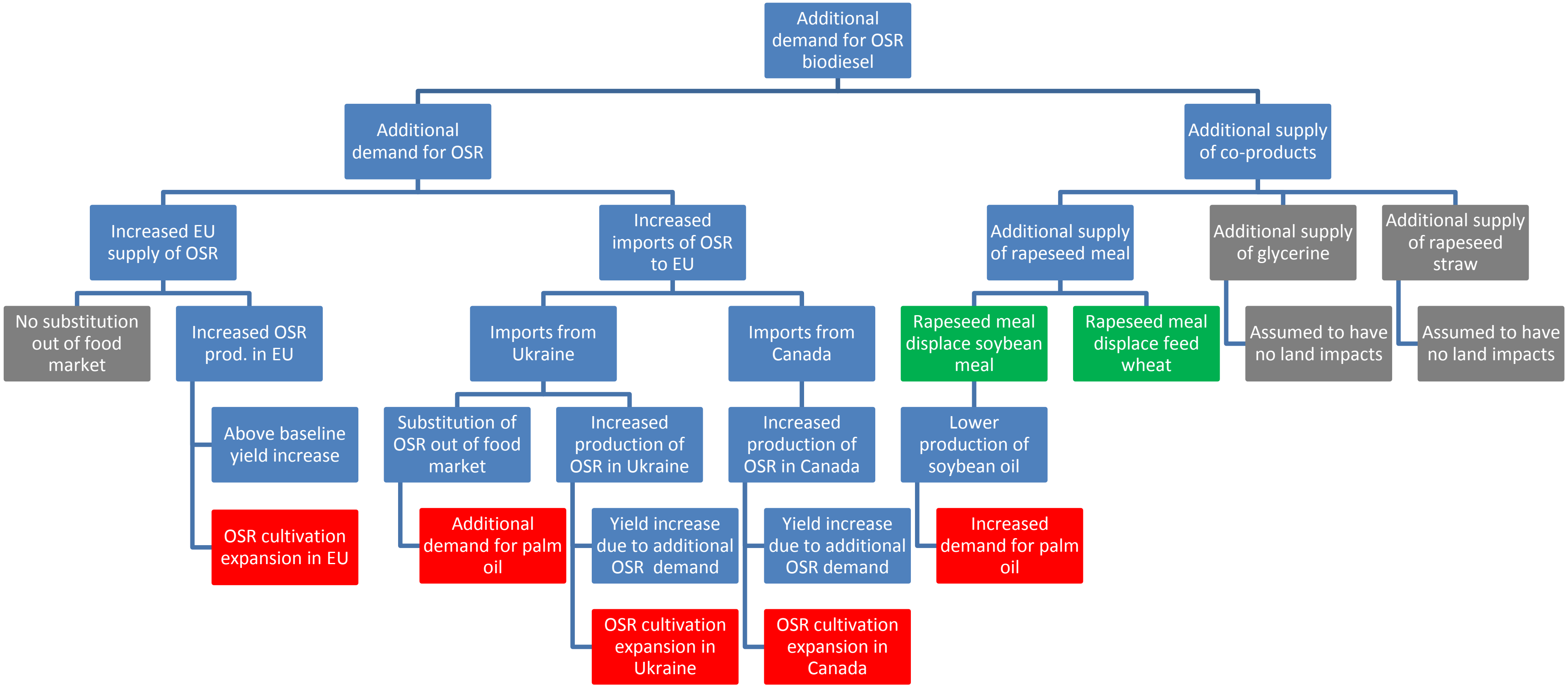

Figure 10. Market responses to an increase in demand for oilseed rape biodiesel.

Land expansion, or ILUC “debits” are shaded red and avoided land expansion, or ILUC “credits” are shaded green. Market responses depicted in the diagram in grey are not considered further for reasons explained in the text.

5.3.1 Increased supply of European oilseed rape

There are three main ways in which the supply of rapeseed oil in Europe could increase (see Figure 10):

- Some rapeseed oil could be displaced out of European food markets and into biodiesel production. The rapeseed oil would then be replaced by some other types of oil in the food market.
- Europe could see an increase in the amount of oilseed rape it produces, as an additional demand for rapeseed oil would push up prices and thus provide an incentive for farmers to plant oilseed rape.
- The trade balance of oilseed rape or rapeseed oil could change. Trade balance change can be due to a decrease in European exports – which would force main importing countries to buy their oilseed rape or rapeseed oil from other exporting countries – and/or to an increase in European imports.

Each of these three points will be discussed in the following sub-sections.

5.3.1.1 Displacement of rapeseed oil out of the European food market

The market share of rapeseed oil in the European food market has remained relatively stable between 1991 and 2009 – varying from a minimum of 18% in 1993 and 1994 to a maximum of 27% in 1999. During the same period, the relative use of palm oil has steadily increased from 15% in 1991 to 21% in 2009 while the shares of coconut oil, peanut oil and especially soybean oil have been steadily decreasing over the same period (see Figure 11) (USDA FAS, 2010).

In absolute terms, the total amount of vegetable oils used in the European food market increased from 8 to 13 million tonnes (USDA FAS, 2010).

This observation of historical trends over a period where biodiesel production in Europe has become more and more important and the amount of rapeseed oil in the food market has remained relatively constant has lead us to assume that no rapeseed oil would be displaced out of the European food market and into biodiesel production.

This assumption is supported by the views of the expert advisory group. The general view is that while most vegetable oils are potential substitutes to rapeseed oil, it is unlikely that any substitution will occur:

- While significantly cheaper than rapeseed oil, palm oil use tends to be constrained due to its physical properties (principally the fact it is solid at room temperature). Experts tend to believe that palm oil has already achieved its technical maximum market share in the EU (particularly EU15 countries).
- The use of soy in the EU food industry has been constrained by concerns about GMOs. While these barriers may no longer exist by 2020, price (soy oil tends to trade at similar prices to rapeseed oil) and availability (due to increasing demand in producing countries) are likely to constrain soybean oil's ability to substitute rapeseed oil in Europe.
- The price of sunflower oil is considered a barrier to substitution of rapeseed oil, as is competition for land.

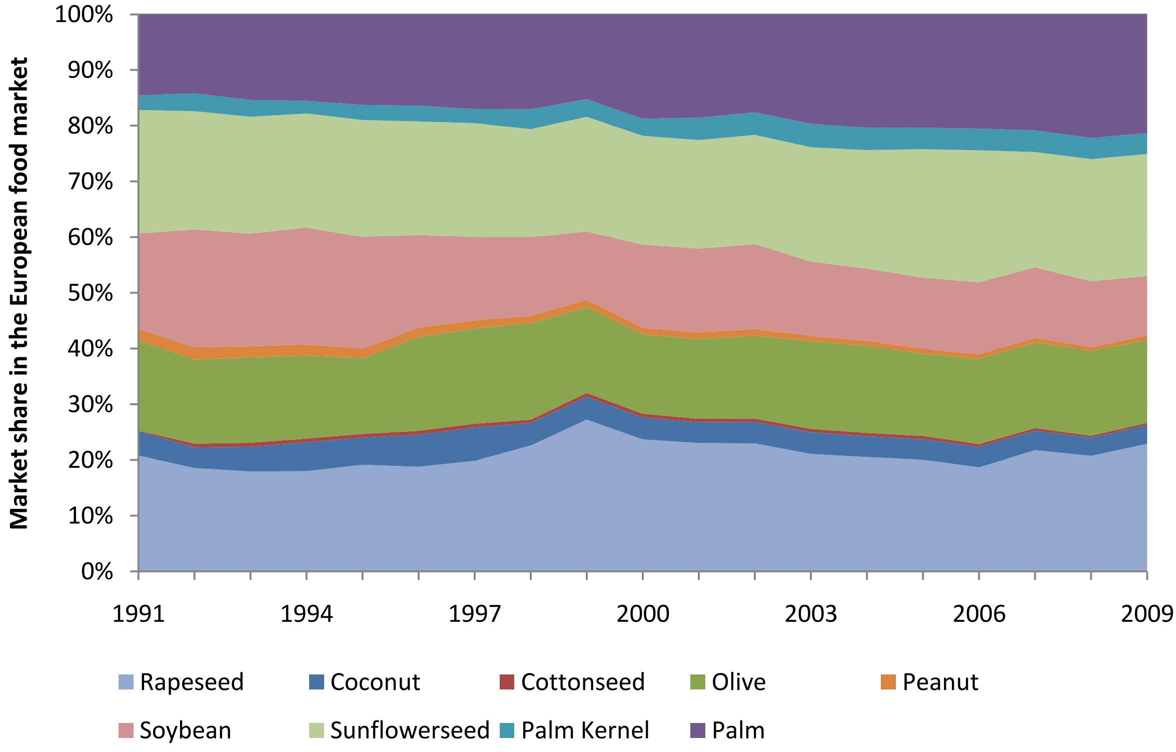

Figure 11. Market share of vegetable oils in the European food market between 1991 and 2009.

Source: USDA FAS (2010)

5.3.1.2 Increased European production of oilseed rape

Yield and area change in Europe. Production of oilseed rape can increase due to improvements in yield and/or increase in harvested area. The projections of yield and cultivation area, both for the baseline and the biofuel projection are based on a method developed by Lywood et al. (2009a) and described in more detail in section 3.3.2.2.

Table 16 below presents the projected oilseed rape area and yield when this method is applied to European oilseed rape cultivation, based on the demand assumptions made in section 5.2.

Table 16. Overview of production, area and yield projections for oilseed rape in Europe in the baseline and biofuel projections in 2008 and 2020.

Projection:	**Baseline**		**Biofuel**	**Additional demand for biofuels**
Year:	**2008**	**2020**	**2020**	**2020**
Production ['000 tonnes]	19,100	19,100	26,264	7,164
Area ['000 ha]	6,250	6,250	8,572	2,322
Yield [tonne / ha]	3.06	3.06	3.06	-

Yield change. In both our baseline and biofuel projection, the yield of oilseed rape stays constant at 3.06 t/ha. This was felt to be reasonable by the expert advisory group as the next ten years will see the phase out of some pest management products that have been beneficial to OSR, and their phasing out could offset likely incremental yield improvements.

Land availability in Europe. While the Lywood et al. (2009a) methodology predicts an increase in oilseed rape cultivation area from 6.3 to 8.6 million hectares, the potential for such an increase in production needs to be ascertained in terms of land availability. Oilseed rape is a break crop, usually grown in rotation with cereals (mainly wheat) in North Western Europe. Break crops are grown for agronomic reasons, such as increased cereal yields, avoided build up of pathogens and pests that often occur when one crop is continuously grown, etc. However, the number and type of cereals and break crops grown within a rotation varies, based on maximising gross margins at the farm.

There are three ways in which oilseed rape harvested area could increase:

- If cereal cultivation areas were expanding, oilseed rape cultivation areas could also expand. However, based on data from FAPRI (2009a) and E4tech's estimated additional demand for cereals for biofuel production, European cereal cultivation areas will be decreasing both in the baseline and biofuel projections (see Figure 12).
- The rotation rate of oilseed rape within a cereal-break crop rotation can increase. However, rotation rates depend on local climate conditions and gross margins. For example, the rotations currently practised in the northern part of Germany are normally three-course rotations with oilseed rape-winter wheat-winter barley or even two-course rotations. In the UK, oilseed rape is normally grown every four years, while in Denmark and Sweden rotational breaks tend to be longer compared with the UK and Germany (Christen et al., 1999). It is usually considered that European rotation rates are high and thus unlikely to get higher (JEC, 2006).
- Oilseed rape can replace other break crops either because these have been decreasing historically or because oilseed rape is economically a better option:
    - On the one hand, there are a number of break crops whose harvested areas have been decreasing over the period 2000-2008 in Europe (see Figure 13). The historical decline in these crops was assumed to continue to 2020, and the freed up land between 2008 and 2020 was considered available for oilseed rape production. This amounted to about 1.1 million hectares.
    - On the other hand, compared to other break crops, oilseed rape is one of the most economically interesting crops. As demand for oilseed rape increases, farmers will be more likely to switch to oilseed rape. Based on gross margins from Nix (2007), we assumed that oilseed rape would replace all flax fibre and tow, linseed, lupins, dry peas and soybeans in Europe[11]. This gave an additional 0.1 million hectares for oilseed rape cultivation.

Through the approach described above, a total availability of land for oilseed rape cultivation in Europe in the biofuels projection was estimated at 1.2 million hectares.

[11] Strictly speaking, if only gross margins from Nix (2007) were taken into account, oilseed rape would also replace sunflower. However, as demand for sunflower oil is also likely to grow, and as sunflower and oilseed rape have always been in direct competition for land, it was felt unlikely that oilseed rape would have displaced all sunflower by 2020.

We recognise the high level of uncertainty associated with this estimation. Further work would be needed, for example through a better modelling of the economics at farm level, to take into account farming practices such as crop rotations. However, our estimate is within the range cited by JEC (2006) of 1 to 2 million hectares of land availability for oilseed rape cultivation.

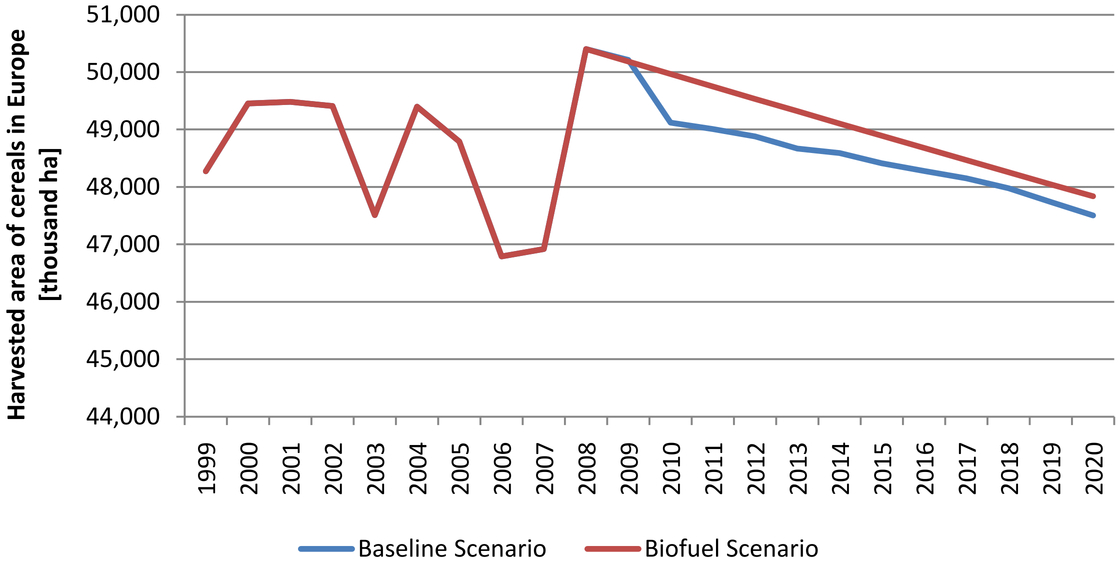

Figure 12. Historical trend and projections in harvested area of cereals in Europe for the baseline and the biofuels projection.

The baseline projection is based on FAPRI (2009a) projections, having withdrawn any additional cereal production for bioethanol production after 2008. The biofuel projection is composed of the baseline demand for cereals plus our projected additional demand due to increased demand for bioethanol.

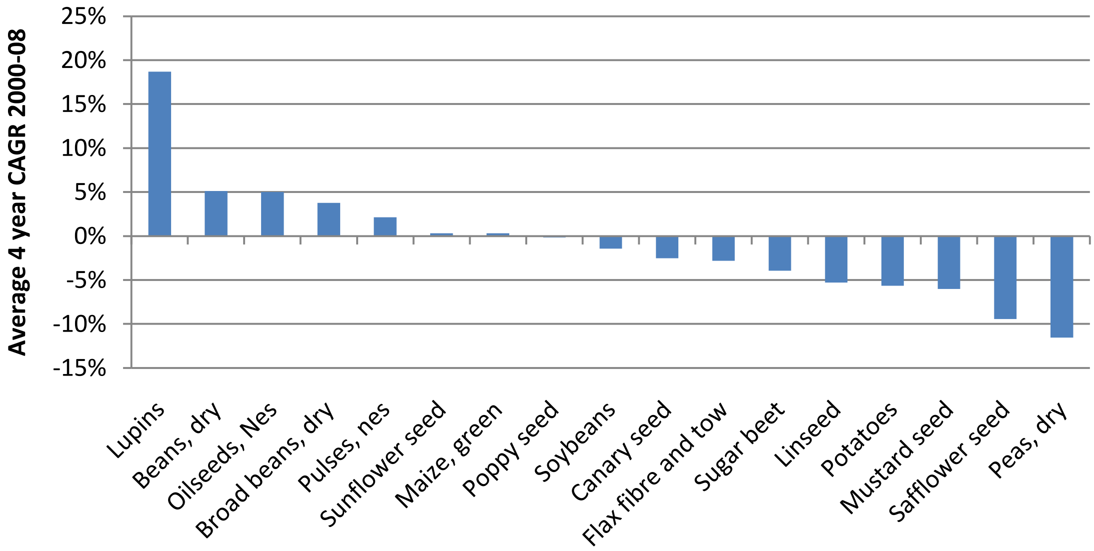

Figure 13. Growth rates in break crops which oilseed rape could potentially displace.

Source: FAOSTAT (2010a).

Actual oilseed rape production in Europe. As can be seen in Table 16, a land availability of 1.2 million hectares is not enough to cover the total demand for oilseed rape in Europe. At a yield of 3.06 t/ha, 1.2 million hectares will produce about 52% of the European demand.

Furthermore, there is clear uncertainty over the 1.2 million hectares estimate, in terms of whether it could become available and whether it would be used for oilseed rape. An assessment of how much of the available land is likely to be used for oilseed rape cultivation would require comparison of production cost of oilseed rape in Europe and prediction of price of oilseed rape. We have not conducted such an analysis. However, we examined a scenario in which only 26% of the oilseed rape would be produced domestically (scenario 6 in Table 15), to test the sensitivity of the ILUC impact to European oilseed rape production.

5.3.1.3 Change in the oilseed rape trade balance due to increased imports

As shown in the previous sub-section, we have considered that Europe will only produce from 26% (scenario 6) to 52% (scenarios 1 to 5) of its own demand for oilseed rape for biodiesel production. The rest of the demand will come from a change in the trade balance of oilseed rape. In this study, we have assumed that the rest of the demand will come from an increase in imports of oilseed rape to Europe.

The main difference between scenarios 1 to 5 and scenario 6 (apart from the amount of imports) is the origin of the imported rapeseed and the form under which it is imported to Europe:

- In scenarios 1 to 5, 48% of the needed rapeseed is imported from Ukraine as oilseed rape.
- In scenario 6, half of the 72% of the needed rapeseed is imported from Ukraine as oilseed rape and the other half from Canada as rapeseed oil.

Ukraine and Canada were considered as they are and are expected to remain the main oilseed rape and rapeseed oil exporting countries (FAPRI, 2009b).

5.3.2 Increased supply of non-European oilseed rape

As determined in sub-section 5.3.1.3, the increased European imports of oilseed rape will come from Ukraine or Canada. In the following sub-sections, the additional supply of oilseed rape in these two countries is analysed.

5.3.2.1 Increased exports from Ukraine to Europe

Displacement of oilseed rape out of the Ukrainian food market. As can be seen from Figure 14, the use of vegetable oil in the Ukrainian food market is quite different from the use in the European market (Figure 11). In Ukraine, sunflower oil is the main vegetable oil, followed by rapeseed and soybean oils, although recent years have seen a relative increase in the share of rapeseed and soy oil. Furthermore, in absolute values, the use of vegetable oil appears to have been decreasing dramatically, from 1.3 million tonnes of oils in 1988 to just under 0.5 million tonnes in 2010 (USDA FAS, 2010).

No data was available to this study on which to base a potential displacement of rapeseed oil out of the food market, as a result of changes in demand and supply. Therefore, we also included a scenario which considers that 50% of the rapeseed oil used in the food market in Ukraine would be diverted into exports to Europe (scenario 2 in Table 15).

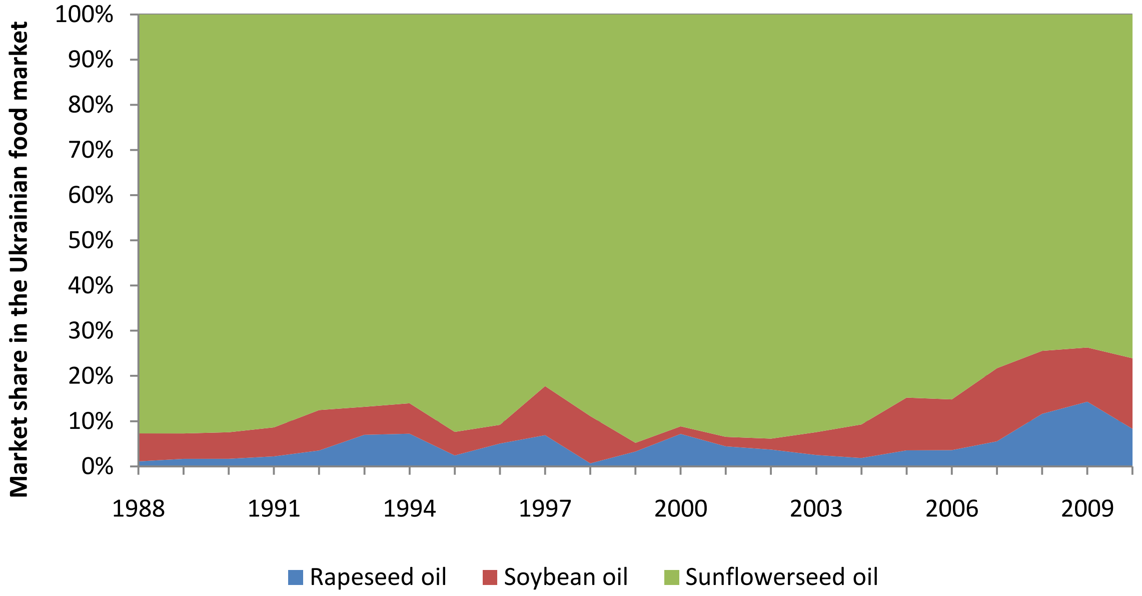

Figure 14. Market share of vegetable oils in the Ukrainian food market between 1988 and 2010.

Source: USDA FAS (2010).

Increased Ukrainian production of oilseed rape. The increase in Ukrainian oilseed rape production is expected to supply most of the export to Europe. To determine the increase of oilseed rape harvested area in Ukraine, we have used the same method as for Europe (described in sub-section 5.3.1.2). Table 17 summarises the production, area and yield projections made for Ukraine in the different scenarios.

Table 17. Overview of production, area and yield projections for oilseed rape in Ukraine in the baseline and biofuel projections in 2008 and 2020.

Scenario:	**All**		**Scenario 1 and 3 to 5**		**Scenario 2**		**Scenario 6**	
Projection:	**Baseline**		**Biofuels**		**Biofuels**		**Biofuels**	
Year:	**2008**	**2020**	**2020**	**Extra demand for biofuels**	**2020**	**Extra demand for biofuels**	**2020**	**Extra demand for biofuels**
Production ['000 t]	2,800	4,078	7,539	3,461	7,506	3,428	6,734	2,656
Area ['000 ha]	1,400	1,984	2,660	676.2	2,655	671.5	2,540	555.9
Yield [t / ha]	2.00	2.06	2.83	-	2.83	n/a	2.65	-

Yield increase. The model predicts an increase in Ukrainian oilseed rape yield from 2.00 t / ha in 2008 to a maximum of 2.83 t / ha in 2020. This corresponds to an annual increase of 2.9%. This value can for example be compared with the CAGR of oilseed rape yield over the previous 12-year period, i.e. from 1996 to 2008, which was of 8.7% (FAO, 2010a). Thus the increase in yield predicted by our model was considered in a feasible range.

Land availability in Ukraine. As oilseed rape is a break crop (with gross margins significantly lower than cereals, Nix (2007)), it is unlikely that oilseed rape would expand directly onto land not used for agriculture. As for Europe, it is more likely that oilseed rape will displace cereals or other break crops out of cereal-break crop rotations. The exact displacement will depend on how close the farmer is to the economic optimum rotation rate of cereals and break crops (even though break crops have lower margins, they help keep the cereal yields high and so farmers see an economic incentive in growing break crops and cereals in turn until a certain optimum is reach). The displaced crops would then have to be produced on “new land”. Determining the amount of land that these crops would expand onto (assuming that the production amount stays constant) depends on the cereal-break crop rotation rate. However this depends on many factors ranging from geographical location to price changes and local gross margin optimisation. We have assumed that the area of crops expansion onto non-agricultural land would be equal to the area expansion due to oilseed rape production[12]. Thus, depending on the scenarios, we would see a cereal expansion of 676,200 to 555,900 hectares in Ukraine.

With the fall of the Soviet Union in the early 1990s, large amounts of agricultural lands were abandoned in Ukraine (Vuichard et al., 2009). The Food and Agriculture Organisation estimated that about 20 million hectares of agricultural land was abandoned during the 1990s in the former Soviet Union (FAO, 2004). We have therefore assumed that there would be enough land available in Ukraine for such a cereal area expansion.

5.3.2.2 Increased exports from Canada to Europe

Canada is currently the biggest exporter of rapeseed products and supplies rapeseed oil and/or oilseed rape to China, the USA, Mexico but its current export to Europe is low. However, in scenario 6, i.e. where the demand for rapeseed oil is high and Europe only produces 26% of its demand, it was assumed that Europe would increase its imports (compared to the baseline) both from the Ukraine (as shown in the previous sub-section) and from Canada.

It is furthermore assumed that Canada will supply Europe with rapeseed oil through increasing its domestic production. Thus no increase in imports and no displacement of rapeseed oil out of the Canadian food market are modelled.

For production in Europe and Ukraine, we use the Lywood et al. (2009a) methodology to determine the expected area and yield increase due to the additional demand for biofuels (see sub-sections 5.3.1.2 and 5.3.2.1). However, as Lywood et al. (2009a) did not consider Canadian oilseed rape production in their paper, we have based our estimates of the contribution of yield and area increase to the increase in demand directly on historical data. The results can be seen in Table 18 below.

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[12] If marginal yields were used, determining the exact crop and its yield would have been important. Indeed, if oilseed rape harvested area was to increase by 1 ha, it would displace 1 ha of wheat onto a land on which wheat would have a lower yield. Thus, 1 ha of oilseed rape increase would be equivalent to 1.2 ha of wheat cultivation (numbers used as examples only). However, we have used average yields in this study and we have thus not considered this effect. For a justification of this choice, please read section 3.3.2.2, page 20.

Table 18. Overview of production, area and yield projections for oilseed rape in Canada in the baseline and biofuel projections (scenario 6 only) in 2008 and 2020.

Scenario:	Scenario 6			
Projection:	**Baseline**		**Biofuels**	
Year:	**2008**	**2020**	**2020**	**Additional demand due to biofuels**
Production ['000 tonnes]	12,600	13,423	16,080	2,656
Area ['000 ha]	6,490	6,731	7,428	697.4
Yield [t / ha]	1.94	1.99	2.16	-

5.3.3 Effect of co-products

The production of biodiesel from oilseed rape has two main co-products: rapeseed meal and glycerine. Just as for palm, glycerine was not considered to have any land impacts and was thus not included in the model (see section 4.3.5.3). Furthermore, the cultivation of oilseed rape also produces rapeseed straw. However, for this study we have considered that rapeseed straw does not displace any land-grown products. We assumed that rapeseed meal would have a bigger impact on the ILUC factor than rapeseed straw, and thus concentrated our co-product analysis on the meal. The effect of rapeseed straw could be included in further future analysis to understand its effect.

Thus the only co-product with land impacts is assumed to be rapeseed meal. Rapeseed meal is used as an animal feed (Lywood et al., 2009b; Schmidt and Weidema, 2008), and is assumed to replace soybean meal and feed wheat. The approach taken to estimate the land impact of soybean meal and feed wheat substitution is described in section 4.3.5.

Based on stakeholder feedback, two assumptions in this approach were highlighted as potentially important in influencing the ILUC factor. First, the share of rapeseed meal that would actually be used for animal fodder was discussed. Domestic production of protein fodder cannot cover the European demand, which leads to important imports of meals such as soybean meal. So there should certainly be a market for the rapeseed meal in Europe. However, as more protein meal becomes available due to biofuel production, less of it may find use as animal feed. Furthermore, it may become an attractive economically to use some rapeseed meal as a feedstock for power generation.

Detailed work on the economics of animal feed market would be required to assess the amount of rapeseed meal that is likely to be used as animal feed. We did not conduct such a model in this analysis. However, we examined the effect of only using 50% of the rapeseed meal for animal fodder in scenario 4 (Table 15) instead of using 100% as in the other scenarios.

The second topic of discussion was the displacement ratio of soybean meal and feed wheat by rapeseed meal. In section 4.3.5, we presented ratios calculated in Lywood et al. (2009b). However, these ratios were criticised for not considering the difference in the protein quality of meals[13], leading to an overestimation of the soybean substitution ratio and an underestimation of the feed wheat substitution ratio. Scenario 5 examines how the change in substitution ratio can influence ILUC factors by using the ratios published by JEC (2008). Table 19 below reproduces these substitution ratios.

Table 19. Substitution ratios for biofuel co-products as calculated by the JEC (2008).

Co-product	t co-product/t feedstock	Displacement ratio of soybean meal (t soybean meal/t co-product)	Displacement ratio of feed wheat (t wheat/t co-product)
Rapeseed meal	0.57	0.382	0.497
Palm kernel expeller[14]	0.03	0.118	0.710
Wheat DDGS[15]	0.33	0.303	0.766
Corn DDGS[15]	0.31	0.262	0.813

5.4 Land use impacts and greenhouse gas consequences

Land use impacts were calculated based on the difference in harvested area between the baseline and the biofuel projection in 2020 and on the land conversions and associated GHG emissions estimated by Winrock International for the U.S. EPA and applied in the Renewable Fuel Standard 2010 (RFS 2). Winrock's approach is described in further detail in section 3.3.4 (U.S. EPA, 2010).

In the following sections we have presented the type of land use change impacts associated with the production of biodiesel from oilseed rape in each of the three producing regions and the GHG consequences. Finally, the land impacts of co-products are presented in the last section.

5.4.1 Land use change from European supply

The amount of land in Europe that is available for oilseed rape production was determined in section 5.3.1.2. Two different types of land were assumed to be available to oilseed rape production:

- A large part of the European land available was due to displacement of break crops with a historically decreasing trend (about 1.1 million hectares). We assumed that in the baseline,

[13] Different parameters are important to take into account when looking into animal diet. The quantity of protein, measured as the energy to protein ratio is important, a high ratio being needed to optimize the use of the protein (FAO, 2002). This ratio can be corrected by feeding the animal more or less energy-rich feed based on the type of meal used.

Furthermore, the quantity and type of amino acids included in proteins (i.e. the quality of the meal) are other important factors, as they are key issues for appropriate protein use by the animal. Rapeseed meal for example contains glucosinolates, which need to be removed for use as animal feed. It also contains less lysine than soybean meal. However, it provides a much higher proportion of sulphur-containing amino acids (cysteine and methionine) (FAO, 2002). Again, the quality of the meal can be corrected by using synthetic amino acids.

[14] Palm kernel expeller (or PKE) is a co-product of the palm oil production (as discussed in section 4.3.5.2).

[15] DDGS stands for Distillers' Dried Grains with Solubles and is a co-product of bioethanol production (as discussed in section 7.3.3).

this land would have been used for cereal cultivation instead of break crop cultivation. In the biofuel projection however, these declining break crops would be replaced by oilseed rape. Thus, the cereals that were grown on the land freed up by the declining break crops in the baseline have to be grown somewhere else in the biofuel projection, displacing other crops. This leads to a series of knock-on land use change effects, as one crop replaces another until the last one expands onto land with not under agricultural use.

However, as in both the baseline and the biofuel projection, the harvested area of cereals in Europe declines (the difference between the wheat, maize and barley harvested area in 2008 and in 2020 in the biofuel projection is a reduction of 2.6 million hectares), the knock-on effect due to additional oilseed rape demand would not cause actual land use change by bringing land into agricultural production, but would prevent some agricultural land from being abandoned. This abandoned agricultural land would have accumulated some carbon stocks, the amount of which depends on the type of land it would have become. Table 20 shows the share of the different types of land that would have replaced abandoned agricultural land in Europe, together with the carbon stock this land would have accumulated over 30 years (i.e. reversion factors).

- The other part of the European land available to oilseed rape cultivation (about 0.1 million hectares) was due to some break crops being totally substituted by oilseed rape. In the baseline, these break crops would not have been displaced. So in this case no actual land use change is recorded (apart from a change in the crop grown).

In scenario 6, a lower oilseed rape cultivation increase in Europe is assumed (0.6 instead of 1.2 million hectares). We have assumed that all of the 0.6 million hectares taken up by oilseed rape cultivation would be part of the first type of land (i.e. would displace cereal cultivation).

Table 20. Types of land that replace abandoned agricultural land and associated reversion factors for Europe.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Reversion data	
		30 year reversion factors	95% confidence interval
Forest	45%	28.8 t CO_2e / ha	± 24 t CO_2e / ha
Grassland	11%	20.9 t CO_2e / ha	± 25 t CO_2e / ha
Mixed	15%	39.4 t CO_2e / ha	± 25 t CO_2e / ha
Savannah	17%	25.5 t CO_2e / ha	± 22 t CO_2e / ha
Shrub land	11%	34.6 t CO_2e / ha	± 23 t CO_2e / ha

95% confidence intervals were combined to calculate the uncertainty ranges around the ILUC factors, due to the uncertainty in carbon stock measurements.

5.4.2 Land use change from Ukrainian supply

5.4.2.1 Land use impacts and associated greenhouse gas consequences of increased oilseed rape production in Ukraine

The amount of land likely to be taken up by oilseed rape production in Ukraine was determined in section 5.3.2.1, and is dependent on the scenario considered.

Again, oilseed rape is considered a break crop, likely to push cereals to expand onto non-agricultural land rather than to expand directly onto non-agricultural land. So, it has been assumed that cereals would expand onto non-agricultural land in the same amount as oilseed rape cultivation area is expanding. Land availability for the expansion of cereals in Ukraine is discussed in section 5.3.2.1.

Type of land expanded onto. In the Ukraine it is assumed that cereals would be expanding onto areas not already under crop production according to the land types and proportions determined by Winrock International's analysis for the RFS2. Table 21 presents the type of land that typically crop/pasture expands onto and the GHG emissions due to the land use change.

Table 21. Types of land use change and associated emission factors for Ukraine. Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year reversion factors	95% confidence interval
Forest	3%	114 t CO_2e / ha	± 5.4 t CO_2e / ha
Grassland	31%	14.5 t CO_2e / ha	± 17 t CO_2e / ha
Mixed	20%	35.0 t CO_2e / ha	± 11 t CO_2e / ha
Savannah	32%	18.5 t CO_2e / ha	± 14 t CO_2e / ha
Shrub land	13%	26.5 t CO_2e / ha	± 12 t CO_2e / ha
Wetland	2%	20.5 t CO_2e / ha	± 13 t CO_2e / ha
Barren	1%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

5.4.2.2 Land use impacts and associated greenhouse gas consequences of displacement of oilseed rape out of the Ukrainian food market

In scenario 2 (see Table 15), 50% of the rapeseed oil used in the Ukrainian food market is diverted out of the food market and into Ukrainian oilseed rape exports to Europe. The diverted oil will have to be replaced by another vegetable oil, which was assumed to be palm oil, as it is the least cost alternative. Furthermore, as palm oil is not currently used in the Ukrainian food market (USDA FAS, 2010), it is thus assumed that palm oil consumption there is not constrained by its technical characteristics.

To assess the indirect land use change impacts of this replacement, the ILUC factor of palm oil, as calculated in section 4, was used. We determined that palm scenario 3 was the best palm scenario for association with oilseed rape scenario 2 (see Table 24 on page 65).

5.4.3 Land use change from Canadian supply

As for Ukraine, oilseed rape is considered unlikely to expand directly onto non-agricultural land in Canada, but oilseed rape expansion will push cereals cultivation to expand to the same extent as oilseed rape.

In Canada, the baseline and biofuels projections both expect cereal area to decline between 2008 and 2020 by 1.6 million hectares (based on projections by FAPRI, 2009a), enough to cover the cereal expansion due to increased European demand for oilseed rape, which was estimated at 0.7 million hectares in section 5.3.2.2. Thus, similarly to the European case, Winrock International's reversion data will be used to assess the foregone sequestration due to increased exports of Canadian oilseed rape to Europe. The Winrock data used is presented in Table 22.

Table 22. Types of land that replace abandoned agricultural land and associated reversion factors for Canada.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Reversion data	
		30 year reversion factors	95% confidence interval
Forest	43%	42.1 t CO_2e / ha	± 15.8 t CO_2e / ha
Grassland	11%	16.2 t CO_2e / ha	± 142 t CO_2e / ha
Mixed	4%	31.7 t CO_2e / ha	± 16.3 t CO_2e / ha
Savannah	15%	19.1 t CO_2e / ha	± 16.6 t CO_2e / ha
Shrub land	26%	25.1 t CO_2e / ha	± 16.0 t CO_2e / ha

5.4.4 Land use change due to co-products

5.4.4.1 Land use impacts and associated greenhouse gas consequences of soybean meal displacement

Avoided land use change in Argentina and Brazil. The displacement of soybean meal means that less soybean will be grown. Major soybean producers are Argentina and Brazil and it is expected that they will see a decrease in demand for their soybean meal and thus for soybeans. The rapeseed meal thus earns a "credit" to the oilseed rape biodiesel because less demand for soybean means reduced soybean cultivation area expansion and so avoided land use change.

Table 23 below presents the type of land soybean cultivation could typically expand onto and the GHG emissions saved through avoiding this land use change, for both Argentina and Brazil.

Table 23. Types of land use change and associated emission factors for Argentina and Brazil. Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year reversion factors	95% confidence interval
Argentina			
Forest	12%	61.0 t CO_2e / ha	± 82 t CO_2e / ha
Grassland	26%	11.2 t CO_2e / ha	± 58 t CO_2e / ha
Mixed	27%	22.6 t CO_2e / ha	± 47 t CO_2e / ha
Savannah	17%	14.4 t CO_2e / ha	± 40 t CO_2e / ha
Shrub land	14%	20.6 t CO_2e / ha	± 41 t CO_2e / ha
Wetland	1%	15.9 t CO_2e / ha	± 40 t CO_2e / ha
Barren	3%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha
Brazil			
Forest	19%	131 t CO_2e / ha	± 108 t CO_2e / ha
Grassland	18%	30.6 t CO_2e / ha	± 18.2 t CO_2e / ha
Mixed	20%	57.6 t CO_2e / ha	± 30.8 t CO_2e / ha
Savannah	35%	39.7 t CO_2e / ha	± 15.9 t CO_2e / ha
Shrub land	6%	58.9 t CO_2e / ha	± 14.1 t CO_2e / ha
Wetland	0%	44.7 t CO_2e / ha	± 16.9 t CO_2e / ha
Barren	0%	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha

Consequences of displaced soybean meal on vegetable oils. The decrease in soybean production means that less soybean oil (co-product of the soybean meal production) will be produced. This soybean oil will have to be replaced by other vegetable oils. We have considered that palm oil will be replacing soybean oil as the least cost alternative.

To account for the indirect land use change impacts of the increase in palm oil production, we have used the ILUC factor for palm oil as calculated in section 4 of this report. Table 24 below summarises which palm oil scenarios were used in each of the oilseed rape scenarios.

Most oilseed rape scenarios are linked to palm scenario 3. Parameters in palm scenario 3 are:

- No palm yield increase attributed to the additional demand for palm biodiesel. This was assumed as it offers most sensitivity variations in the palm chain.
- Single plantation lifetime. This corresponds to the assumption in the oilseed rape biodiesel chain that the land use will stay the same for 30 years.
- Expansion onto 5% peatland.
- Historical deforestation rates. This corresponds to the assumption in the oilseed rape biodiesel chain to consider historical deforestation rates as well.

However, Oilseed rape scenario 4 corresponds to testing the influence of forest protection policies on the ILUC factor for oilseed rape biodiesel. This was done by lowering the deforestation rate in Indonesia and Malaysia from the historical trend to 10%. Of course this was linked to higher conversion rates of other land types. The palm scenario that best represented this assumption is palm scenario 6.

Table 24. Matching palm and oilseed rape scenarios.

Oilseed rape scenario	1	2	3	4	5	6
Palm scenario	3	3	3	6	3	3

5.4.4.2 Land use impacts and associated greenhouse gas consequences of feed wheat displacement

For feed wheat the situation is quite different. Rapeseed meal is likely to displace domestically produced feed wheat as both Europe and Canada are major wheat producers. We have thus differentiated the credit the oilseed rape ILUC factor receives for displacing feed wheat based on its actual production location.

Displacement of European feed wheat. The displacement of European-grown feed wheat by European rapeseed meal (i.e. meal produced in Europe) leads to a lower demand for wheat in Europe, and thus to lower production. A lower wheat production in Europe is assumed to lead to more agricultural land being abandoned (see section 7.3.1). Thus the credit for displacing feed wheat is calculated using the reversion data from Winrock International (US EPA, 2010). Table 20 on page 60 shows this data for Europe.

Displacement of Canadian feed wheat. The displacement of Canadian-grown feed wheat by rapeseed meal produced in Canada is assumed to lead to a lower demand for wheat in Canada. A lower wheat production in Canada leads to more agricultural land being abandoned (as wheat cultivation areas in Canada are decreasing, see section 7.3.2.2). Thus the credit for displacing feed wheat is calculated using the reversion data from Winrock International as shown in Table 22 on page 63.

5.4.4.3 Greenhouse gas consequences of rapeseed meal not used as animal feed

In scenario 3, we have considered that only 50% of the produced rapeseed meal would be used as animal feed. A likely other use for rapeseed meal is co-firing in power plants to produce electricity. We have considered that the ILUC factor should not receive a credit for this alternative use, as there would be no ILUC impacts associated with displacing coal, and the emissions saved from co-firing would be credited to the calculation of the "direct" GHG emissions.

5.5 Scenario results

Figure 15 presents the ILUC factor for each for the oilseed rape scenarios. As expected, some assumptions have bigger impacts than others on the results. Assumptions leading to high ILUC factors include a low utilisation of rapeseed meal as animal fodder (scenario 3), a low displacement rate of soybean meal by rapeseed meal but a higher displacement rate of feed wheat (scenario 5) and a low production of oilseed rape in Europe with higher productions in Ukraine and Canada (scenario 6).

Other assumptions lead to lower ILUC factors, such as for example good (i.e. effective) anti-deforestation policies in Malaysia and Indonesia (scenario 4) and high European oilseed rape production (scenario 1 and 2).

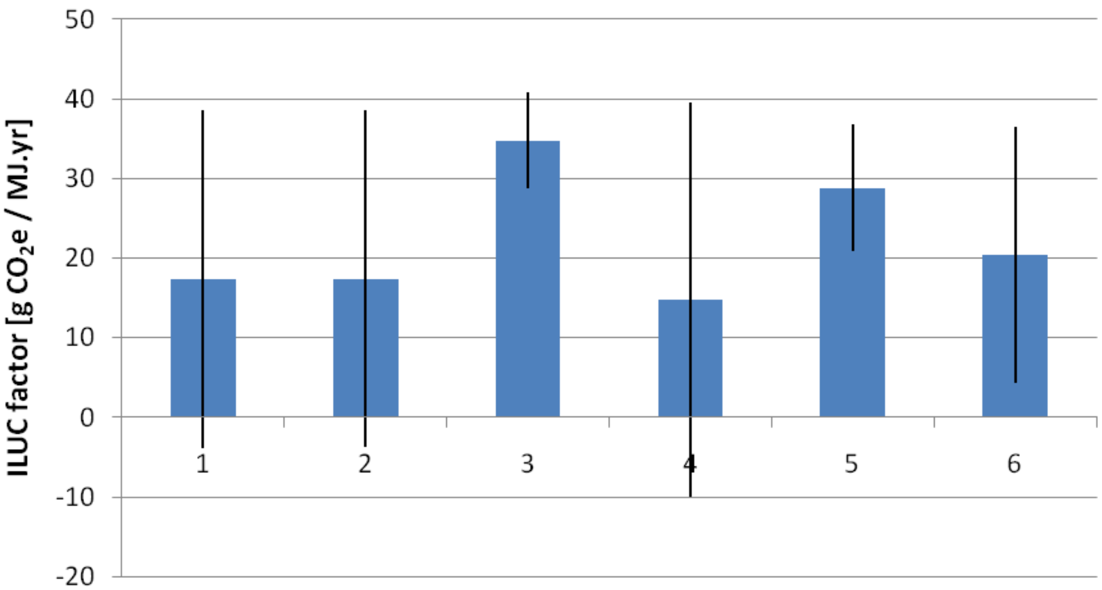

Figure 15. Indirect land use change impacts for the different scenarios modelled for oilseed rape biodiesel.

The uncertainty bars included in Figure 15 represent only the uncertainty associated with carbon stock values. As we can see, these uncertainties are large – some bigger than the ILUC factor itself. This points to a significant need for improved estimations of carbon stocks before any quantitative conclusions can be drawn on indirect land use change impacts.

Figure 16 details the contribution of each market response to the final ILUC factor for one of the scenarios. We have used scenario 6 as an example, the data for the other scenarios can be found in Annex 4. As can be seen, the ILUC factor is composed of debits, i.e. actual land impacts, and of credits, i.e. avoided land impacts. All credits come from the co-products. However debits come both from the cultivation of the biofuel feedstock, in this case oilseed rape, and from other impacts, such as the increased production of palm oil to replace the soybean oil no longer produced due to the lower demand for soybeans due to soybean meal displacement by rapeseed meal.

The waterfall diagram highlights some important parameters that have an impact on the magnitude of ILUC:

- Ukraine and Canada are assumed to provide the same amount of oilseed rape to Europe. However, the GHG impact of expansion in Canada is bigger than for the Ukraine. This is due to (a) a difference in yield (Canada has lower yields so needs more land to produce the same amount of oilseed rape) and (b) the type of land use caused by oilseed rape production (Canada is a highly forested country whereas Ukraine has more grassland).
- The same amount of soybean meal is displaced in Argentina and Brazil. However, the GHG credit for Brazil is much bigger. This is due to the preservation of a significantly higher carbon stock of Brazilian forest compared to Argentinean forest, combined with a higher deforestation rate in Brazil than in Argentina.

- EU feed wheat displacement is only given a very small credit. This is due to the fact that wheat has a very high yield in Europe. Thus displacing even important amounts of EU-grown feed wheat leads to only a small amount of land being "freed" from agricultural use.

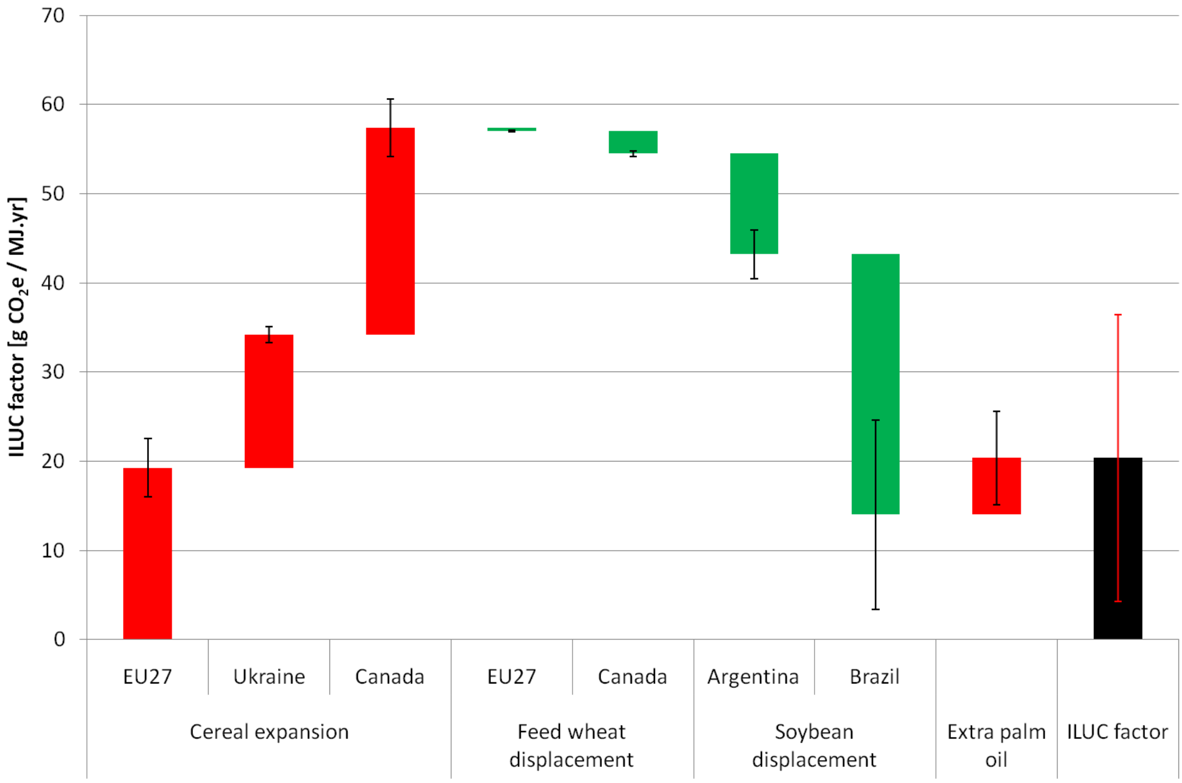

Figure 16. Waterfall diagram showing the contribution of each market response to the overall ILUC factor for scenario 6.

As can also be seen from Figure 16, accounting for the effect of co-products is highly important, as this reduces drastically the ILUC impact of biofuels. Results from scenario 3 also show that it is important consider the use of the co-products, and calculate the credit given to the biofuel based on this use.

6 Soy biodiesel

6.1 Introduction

Soybean biodiesel has made up a significant proportion of global biodiesel consumption in recent years. FAPRI data (2009) suggests that in 2008, 4 million tonnes of soybean oil were used for biodiesel. However, unlike the other biofuels considered in this study, soybean oil is a dependent co-product, rather than a determining co-product, and this makes the results of the analysis rather different to the other chains.

This chapter starts with a description of how the additional demand for soybean oil in 2020 was estimated. It then goes on to discuss in more detail in section 6.3 the market responses to that increased demand. Section 6.4 discusses the land use changes that are modelled to take place and the GHG impacts of those land changes. Section 6.5 presents and discusses the results of this chain. The final section outlines particular aspects of the analysis that could be studied in more detail by those developing further analysis of this kind.

Scenarios. For soybean biodiesel, three scenarios were explored, which are outlined in Table 25. As the only market effect that has an effect on the ILUC impact (as modelled here) is its replacement in the food market by other vegetable oils, these scenarios explore the ILUC impacts through substituting soybean oil with different proportions of vegetable oils (with different ILUC impacts associated with them). Scenarios 1 and 2 assume equal substitution of soybean oil by rapeseed oil and palm oil in the Chinese market, whereas scenario 3 assumes much less (10%) palm oil can be used to substitute into the Chinese markets. Scenarios 1 and 2 differ in terms of the ILUC impacts associated with the substituting palm oil in all markets; scenario 1 assumes that soy oil is substituted by palm oil with high ILUC impacts and scenario 2 assumes it is substituted by palm oil with low ILUC impacts.

Table 25. Scenarios explored for soybean oil biodiesel.

Scenario	1	2	3
Soy oil is substituted in China by 50% rapeseed oil and 50% palm oil	✓	✓	X
Soy oil is substituted in China by 90% rapeseed oil and 10% palm oil	X	X	✓
Soy oil that is substituted by palm oil with high ILUC impacts (palm oil scenario 4)	✓	X	✓
Soy oil that is substituted by palm oil with low ILUC impacts (palm oil scenario 8)	X	✓	X

6.2 Additional global demand for soybean biodiesel in 2020

Estimation of the amount of soybean oil required in 2020 was estimated as described in the first paragraph of section 3.2. Current volumes of soybean oil used for biodiesel (FAPRI, 2009) were subtracted from these projected volumes to provide projections of additional soybean oil demand for biodiesel in 2020. Further information on the biofuel projection is described in section 3.2.

In summary, an additional 10.3 million tonnes soybean oil (above 2008 consumption) is estimated to be used in 2020 for soybean oil biodiesel.

6.3 Market responses

The top three exporting countries of soybean oil are Argentina, Brazil and USA (FAO Statistics, 2005) and in this analysis, this situation is expected to continue to 2020.

As with the other fuel chains considered, the key market responses to increased demand for soybean oil biodiesel were explored. However, as mentioned above, soybean oil is considered in this analysis as a dependent co-product, so biofuel-related market effects have limited influence on LUC impacts of soybean, and the dominant effect is the diversion of soybean oil from food markets. Similarly, it is not appropriate to consider soybean oil co-products as ILUC credits.

These market responses to the increased demand for soybean oil for biodiesel are mapped out in Figure 17, with a focus on the largest soybean oil consuming countries. The magnitude of the effect of the market responses are discussed further in this chapter and illustrated through the scenarios.

As for the other fuel chains, increased soybean oil production through improvements in efficiency yields in the supply chains and reduction in demand for vegetable oils in other markets have not been considered in this analysis. The extent to which these market effects make a contribution could be explored in future analysis of this type.

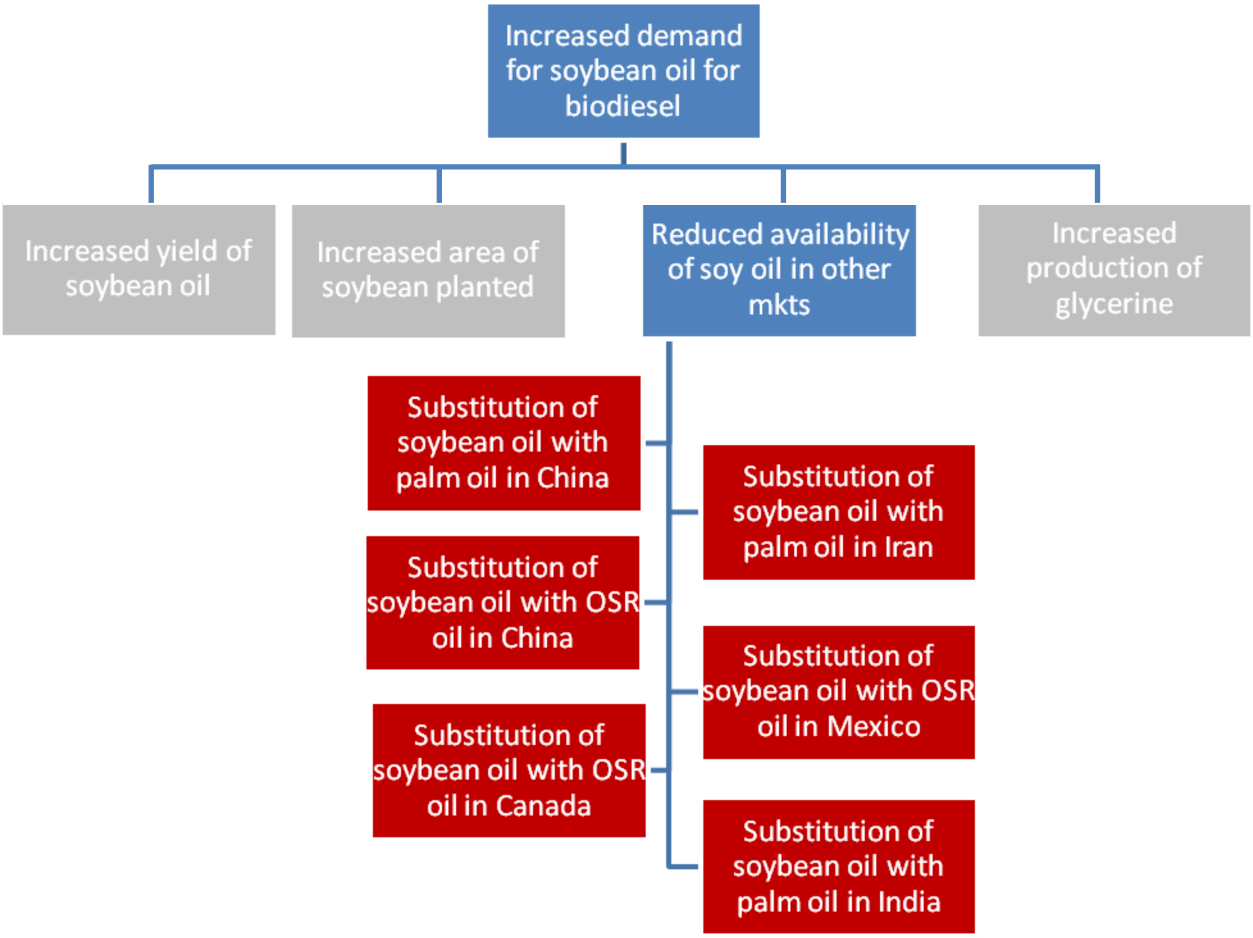

Figure 17. Market responses to an increase in demand for soybean oil for biodiesel.

Land expansion, or ILUC "debits" are shaded red. There are no ILUC "credits" shown explicitly in this diagram, although these will be included intrinsically in the ILUC impacts calculated for the substituting vegetable oils. Market responses depicted in the diagram in grey are not considered further for reasons explained in the text.

6.3.1 Substitution

All the indications are that soybean area grown is only influenced by demand for soybean meal, i.e. within certain limits an increase in price for soy oil will not lead to an increase in the area of soybeans grown. As noted by the Brazilian oilseed processors association ""*It is a mistake to believe that the private sector will make decisions based on just 1/5 of the product* [i.e. the oil]*, without a defined market for the other 4/5* [i.e. the meal]" (ABIOVE, 2009).

However, there is evidence that suggests that there may be a very small elasticity of soybean area in response to soybean oil price. However this elasticity is small, i.e. a 1% increase in soy oil price would lead to a 0.06% in soybean area (Personal Communication with INAI, 2010). One stakeholder believed the effect could potentially be larger than this.

However, based on the broad consensus of opinion of those spoken to, the assumption was therefore made that soybean area planted would not be affected by increased demand for soy oil, and consequently that it is unlikely that soy oil yield would be affected either. However, the assumption that neither of these effects contribute to the response to increased demand for soy oil production is a subject that could be looked at in more detail in further analysis.

One interesting point put forward by a stakeholder was that in the past there have been times when there has been no market for soybean oil and it has on many occasions been stored in Argentina as a surplus, and that this surplus has in fact been one of the major drivers behind the formation of the Argentinean soy biodiesel industry. Stockpiling of the soy oil may mean that diversion of the soy oil to the biodiesel market does not necessarily divert soy oil from the food market. The extent and magnitude of the impact of this issue is not discussed further in this analysis but should certainly be looked at in future developments of analysis of this type.

Therefore, the main response to increased soybean oil biodiesel demand in this analysis is assumed to be substitution by other vegetable oils in the key markets that are currently importing soybean oil for non-biofuel use.

In order to model the vegetable oil substitutions that would take place, the domestic consumption of different vegetable oils was explored in the key soybean oil importing countries. The five key soybean oil importing countries were identified as:

1. China
2. India
3. Iran
4. Mexico
5. Canada

Table 26 shows how these countries make up a significant proportion of the exports from the three soybean oil key exporting countries, Argentina, Brazil and USA.

Table 26. Amount of soybean oil imported by key importing countries in 2005.
Source: FAO export statistics, 2005.

Exporting country [EC]	Amount exported, tonnes	Key countries exporting to [IC]	Amount being imported by country	%age of EC's export going to IC
Argentina	4,650,446	China, mainland	1,542,551	33
		India	1,273,148	27
Brazil	2,680,933	Iran	765,558	29
		India	433,529	16
		China, mainland	365,531	14
USA	473,061	Mexico	131,432	28
		Canada	69,985	15

The impact of reduced availability of soybean oil/higher soybean oil prices on these key importing markets was explored by looking at the relative amounts of different vegetable oils consumed domestically in these countries. A hypothesis was then made for each region as to which vegetable oil would be most likely to replace it. Each hypothesis was based on observed trends and patterns in domestic consumption, as illustrated in the graphs. The substitutions are considered in the importing countries rather than the exporting countries, based on the assumption that export markets will meet their own needs first and export the surplus, and it will be import markets that are affected by the reduction in available soy oil.

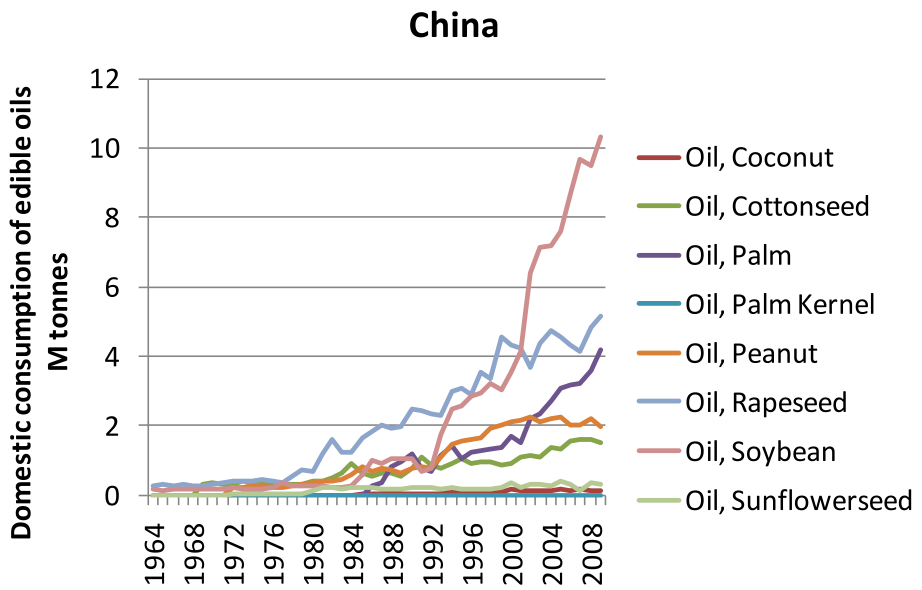

Figure 18. Domestic consumption of edible oils in China.

Source: USDA FAS (2010).

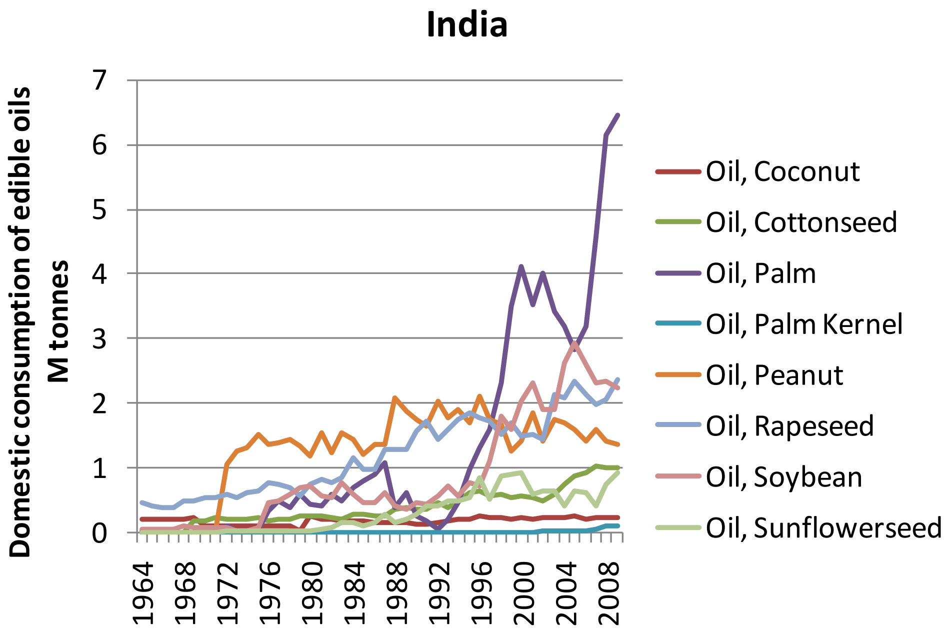

Figure 19. Domestic consumption of edible oils in India.

Source: USDA FAS (2010).

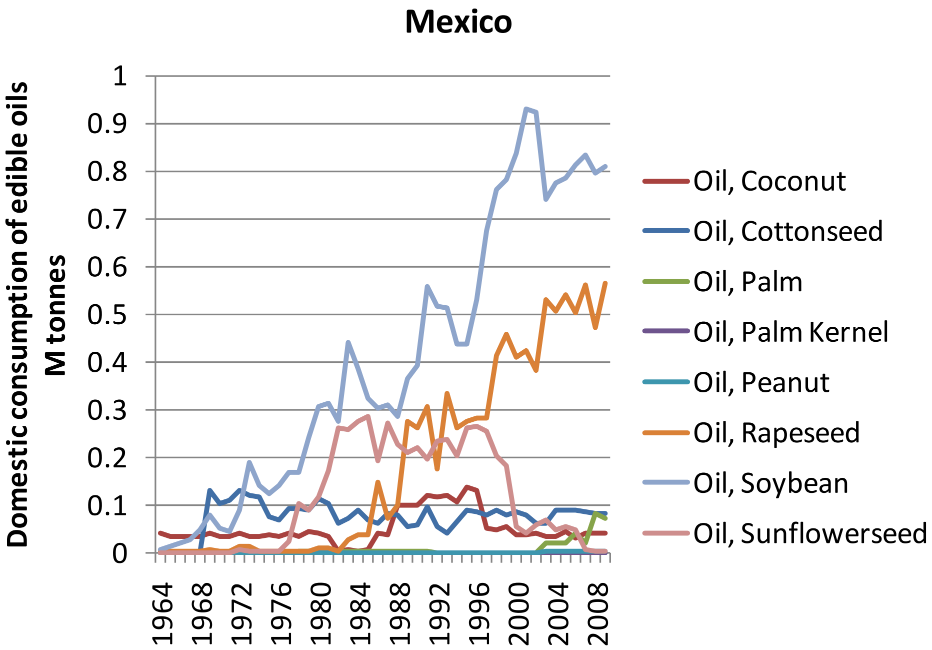

Figure 20. Domestic consumption of edible oils in Mexico.

Source: USDA FAS (2010).

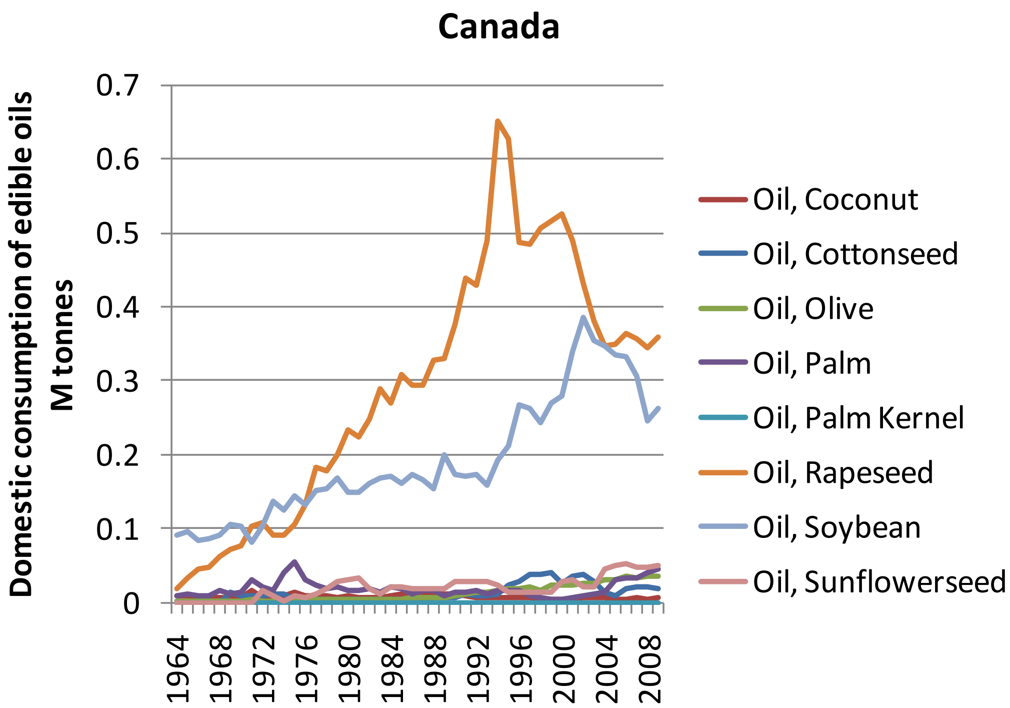

Figure 21. Domestic consumption of edible oils in Canada.

Source: USDA FAS (2010).

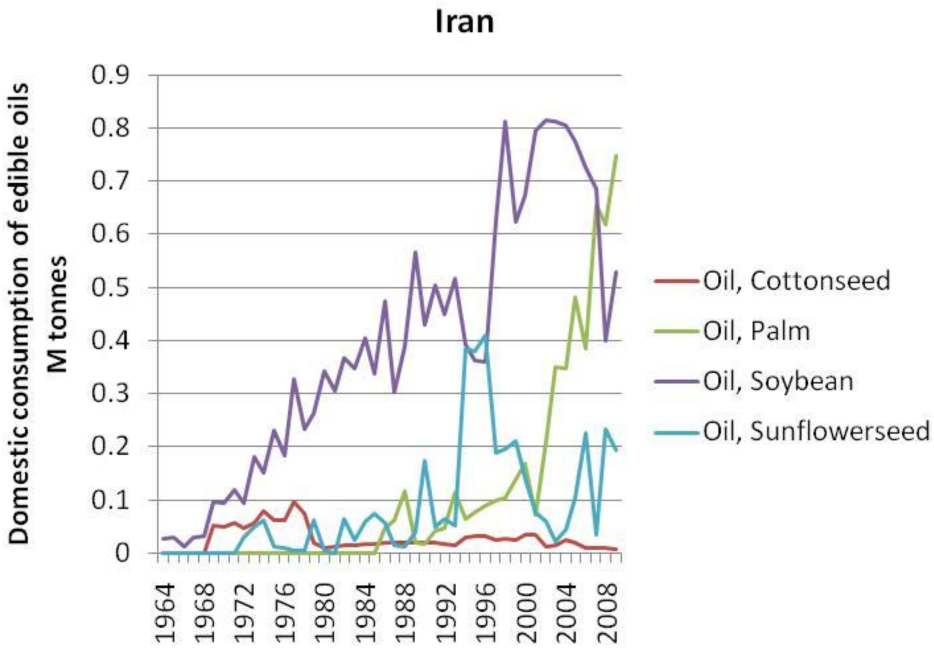

Figure 22. Domestic consumption of edible oils in Iran.

Source: USDA FAS (2010).

The following table shows the substitutions that were assumed in the different scenarios in the different world regions. Scenario 3 assumes a higher substitution of soy oil with rapeseed oil. This is to explore the consequences of a situation in which the Chinese domestic (non-biofuel) market becomes "saturated" in terms of the amount that palm oil can substitute.

Table 27. Product substitutions assumed for different world regions.

Market in which soy oil substituted	Proportion of total soy exports considered going to that country	Substituting product	Proportion product substitutes in scenarios 1&2 [&3]	Country in which substituting product grown
China	42%	Palm	50% [10%]	SE Asia
		OSR	50% [90%]	China & Canada
Mexico	3%	OSR	100%	Canada & US
India	37%	Palm	100%	SE Asia
Canada	2%	OSR	100%	Canada & US
Iran	17%	Palm	100%	SE Asia

There is the possibility that other vegetable oils may become more widely used between now and 2020. For example, corn oil, which can be a by-product of corn bioethanol production, could be an increasing source of oil in the future. As the corn oil would only be produced in the biofuel projection, it would not result in diverting feedstocks out of any current, or baseline, use. Also, because it is a by-product, it could be argued that there are no ILUC impacts associated with its production. If corn oil then substituted for soy oil in some of the above markets, this could lower the overall ILUC impacts of the biofuel projection. This is not explored in more detail here but is certainly an area that could be looked at in more detail in future studies exploring these issues.

6.3.2 Demand induced yield improvements, area increase and co-products

As mentioned earlier in this chapter, increased soy oil prices are not expected to affect soy oil production, as soy oil represents only a fraction of the value of the soy bean. Therefore, the additional demand for soybean biodiesel is not assumed to induce yield increases in soy oil per hectare or increase the area of soy grown. This is an area that could be looked at in more detail in future studies exploring these issues.

Soy oil is considered the dependent co-product of soybean meal production and there are no dependent co-products of soy oil biodiesel production except for glycerine production. As with the other biofuel chains, the ILUC impacts of glycerine substituting other products is not considered but could potentially be an area for future analysis. Therefore no ILUC "credits" are specifically given to any soybean oil co-products. However, when considering that a certain proportion of the soy oil is being substituted by palm oil for example, the ILUC credits associated with the co-products of palm oil production are included in the ILUC impacts associated with substitution by that palm oil.

6.4 Land use impacts and greenhouse gas consequences

6.4.1 Calculation of indirect land use change impacts

The ILUC impacts of all soybean biodiesel used in 2020 are modelled in this analysis by estimating the proportion of the soy oil in other markets that will be replaced by palm oil and rapeseed oil (as these are the two main oils assumed to be replacing soybean oil in the key soybean oil importing markets).

The ILUC impacts (per MJ of vegetable oil produced) calculated in the different scenarios for the other fuel chains are therefore fed into this soybean oil analysis. However, it was also necessary to produce an additional analysis of ILUC impacts associated with rapeseed oil produced only in Canada and the USA combined (as this analysis had not been required for the EU rapeseed biodiesel ILUC factor analysis, and this region was expected to produce much of the rapeseed oil required to substitute soy oil). This additional OSR analysis is described in section 6.4.2.

Table 28 shows the different ILUC impacts that are fed into the 3 different scenarios for soybean biodiesel production.

Table 28. Scenarios from other fuel chains used in soybean biodiesel ILUC factor calculation.

Market in which soy oil substituted	Substituting product	Proportion product substitutes in scenarios 1&2 [&3]	Country in which substituting product grown	Soy scenario 1 - ILUC impacts based on:	Soy scenario 2 - ILUC impacts based on:	Soy scenario 3 - ILUC impacts based on:
China	Palm	50% [10%]	SE Asia	Palm scenario 4	Palm scenario 8	Palm scenario 4
	OSR	50% [90%]	China & Canada	OSR Canada/USA scenario	OSR Canada/USA scenario	OSR Canada/USA scenario
Mexico	OSR	100%	Canada & US	OSR Canada/USA scenario	OSR Canada/USA scenario	OSR Canada/USA scenario
India	Palm	100%	SE Asia	Palm scenario 4	Palm scenario 8	Palm scenario 4
Canada	OSR	100%	Canada & US	OSR Canada/USA scenario	OSR Canada/USA scenario	OSR Canada/USA scenario
Iran	Palm	100%	SE Asia	Palm scenario 4	Palm scenario 8	Palm scenario 4

Palm scenario 4 was used to represent a situation in which soybean oil is replaced by palm oil with high ILUC impacts, and palm scenario 8 to reflect a situation in which soybean oil is replaced by palm oil with low ILUC impacts.

For Mexico and Canada, it was assumed that all of the additional rapeseed oil consumed would have been grown in Canada/USA, as this is the case at present. For simplicity, the situation in China was assumed to be very similar to that in Canada, i.e. that additional OSR demand would be met through expansion of crop area, that relatively similar land type conversions would take place (which is in keeping with the Winrock analysis) and that the emissions factors for these two regions are also broadly similar. How the ILUC factor was estimated is described in the next section.

6.4.2 Indirect land use change impacts of Canadian/USA rapeseed oil

6.4.2.1 Increase in production of oilseed rape in Canada and the USA

Additional demand for oilseed rape in Canada and the USA. The baseline for oilseed rape demand in Canada and the USA is based on FAPRI (2009a) projections. However, to calculate the demand in 2020, the demand for oilseed rape for biodiesel production was kept constant at 2008 levels. This leads to a slight increase in demand for oilseed rape, from 13 to 14 million tonnes in the baseline.

Additional demand for oilseed rape in the biofuel projection is two-fold:

- We have considered that North America will produce biodiesel from rapeseed oil in 2020 (a demand of 0.4 million tonnes of oilseed rape was assumed, see section 3.2).
- Furthermore, as shown in the previous sections, additional demand for oilseed rape should be taken into account for soybean oil substitution. This amounts to about 6 million tonnes of oilseed rape.

Thus the total additional demand for oilseed rape in North America in the biofuel projection is 6.5 million tonnes.

Yield and area projections to 2020. Canada and the USA are together one of the biggest producers of oilseed rape. However, neither Canada nor the USA was considered in the Lywood et al. (2009a) methodological paper. We have therefore entirely based our estimates of the contribution of yield and area increase to the increase in demand directly on historical data. The results can be seen in Table 29 below.

Table 29. Overview of production, area and yield projections for oilseed rape in Canada and the USA in the baseline and biofuel projections in 2008 and 2020.

Projection:	Baseline		Biofuels	
Year:	**2008**	**2020**	**2020**	**Additional demand due to biofuels**
Production ['000 tonnes]	13,255	14,397	20,884	6,487
Area ['000 ha]	6,890	7,225	8,758	1,533
Yield [t / ha]	1.92	1.99	2.38	-

Furthermore, we have assumed that the split in production between Canada and the US in 2020 will remain the same as in 2005, i.e. 93% and 7% respectively.

Type of land used for oilseed rape production. As oilseed rape is a break crop (with gross margins significantly lower than cereals, Nix (2007)), it is unlikely that oilseed rape would expand directly onto land not used for agriculture. It is more likely that oilseed rape, although planted as a break crop, will displace cereals out of a cereal-break crop rotation. Cereals would then have to be produced on "new land". Determining the amount of land that cereals would expand onto (assuming that the production amount stays constant) depends on the amount of cereals that is usually grown within a cereal-break crop rotation. However this depends on many factors ranging from geographical location to price changes and local gross margin optimisation. We have assumed that the area of cereal expansion onto non-agricultural land would be equal to the area expansion due to oilseed rape production[16]. Thus, we would see a cereal expansion of 1.5 million hectares in Canada and the United States.

FAPRI (2009) projects that wheat cultivation area will decline both in Canada and the United States between 2008 and 2020. We have therefore considered that the cereal cultivation area expansion taking place due to increased demand for oilseed rape will not lead to non-agricultural land being taken into production but will rather lead to less of a wheat cultivation area decrease. Thus reversion data from Winrock International is used to assess the foregone sequestration of GHGs due to the additional demand for oilseed rape. Table 30 summarises the Winrock data used for Canada and the United States.

[16] If marginal yields were used, determining the exact crop and its yield would have been important. Indeed, if oilseed rape harvested area was to increase by 1 ha, it would displace 1 ha of wheat onto a land on which wheat would have a lower yield. Thus, 1 ha of oilseed rape increase would be equivalent to 1.2 ha of wheat cultivation (numbers used as examples only). However, we have used average yields in this study and we have thus not considered this effect. For a justification of this choice, please read section 3.3.2.2, page 20.

Table 30. Type of land that abandoned agricultural land would have become and the reversion factors associated with them for Canada and the United States.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Reversion data	
		30 year reversion factors	95% confidence interval
Canada			
Forest	43%	42.1 t CO_2e / ha	± 15.8 t CO_2e / ha
Grassland	11%	16.2 t CO_2e / ha	± 142 t CO_2e / ha
Mixed	4%	31.7 t CO_2e / ha	± 16.3 t CO_2e / ha
Savannah	15%	19.1 t CO_2e / ha	± 16.6 t CO_2e / ha
Shrub land	26%	25.1 t CO_2e / ha	± 16.0 t CO_2e / ha
United States of America			
Forest	27%	64.7 t CO_2e / ha	± 16 t CO_2e / ha
Grassland	31%	11.3 t CO_2e / ha	± 23 t CO_2e / ha
Mixed	14%	37.7 t CO_2e / ha	± 29 t CO_2e / ha
Savannah	12%	15.3 t CO_2e / ha	± 17 t CO_2e / ha
Shrub land	16%	23.3 t CO_2e / ha	± 22 t CO_2e / ha

6.4.2.2 Co-product treatment

The production of rapeseed oil has one main co-product, rapeseed meal. Furthermore, the cultivation of oilseed rape also produces rapeseed straw. However, for this study we have assumed that rapeseed straw does not displace any land-grown products. We assumed that rapeseed meal would have a bigger impact on the ILUC factor than rapeseed straw, and thus concentrated our co-product analysis on the meal. This assumption could be tested further in future analysis.

Thus the only co-product with land impacts is considered to be rapeseed meal in this analysis. Rapeseed meal is used as an animal fodder (Lywood et al., 2009b; Schmidt and Weidema, 2008) and as such replaces soybean meal and feed wheat. The approach taken to estimate the land impact of soybean meal and feed wheat substitution is described in section 4.3.5

Avoided land use change in Argentina and Brazil. The displacement of soybean meal means that less soybean will be grown. Major soybean producers are Argentina and Brazil and it is expected that they will see a decrease in demand for their soybean meal and thus for soybeans. The rapeseed meal thus earns a “credit” to the oilseed rape because less demand for soybean means reduced soybean cultivation area expansion and so avoided land use change.

Table 23 on page 64 presents the type of land soybean cultivation would have expanded onto and the GHG emissions saved through avoiding this land use change, for both Argentina and Brazil.

Consequences of displaced soybean meal on vegetable oils. The decrease in soybean production means that less soybean oil (co-product of the soybean meal production) will be produced. This soybean oil will have to be replaced by other vegetable oils. We have considered that palm oil will be replacing soybean oil as it is the current marginal oil and is expected to remain as such until 2020[17].

To account for the indirect land use change impacts of this increase in palm oil production, we have used the ILUC factor for palm oil as calculated in section 4 of this report (palm scenario 2).

Feed wheat displacement. Rapeseed meal also displaces feed wheat when used as animal fodder. We have assumed that the rapeseed meal produced in this case due to rapeseed oil production would displace feed wheat from the 5 main feed wheat producers (after the EU), i.e. the USA, Russia, Australia, Canada and Argentina.

The land impact of this displacement depends on the trend in wheat cultivation area. As already explained above, USA and Canada have seen their wheat cultivation areas decline historically, and this trend is projected to continue by FAPRI (2009). The same can be said about Russia. So for these countries we used Winrock's reversion data to determine the land types impacted and the GHG consequences of the production of rapeseed meal.

For Argentina and Australia on the other hand the wheat cultivation area is expanding. We have thus assumed that lower feed wheat production would mean avoided land use change, and so we used Winrock's conversion data for these countries.

6.4.2.3 Intermediary results

The ILUC factor calculated through this approach is 17 g CO_2e / MJ of rapeseed oil. This ILUC factor cannot be directly compared to other ILUC factors in this study as the unit is different (MJ of oil in this case instead of MJ of biofuel). This specific unit is used as we are not assessing increased demand for biofuel production but increased demand for rapeseed oil to substitution the soybean oil used in the production of biodiesel.

6.5 Scenario results

The following graph shows the different ILUC factors estimated using our methodology for the different scenarios explored.

[17] This is a simplification. The substitution percentage of soy oil by palm and rapeseed oil as calculated in the previous sections could have been used. However, considering the small amount of soy oil, a first order approximation was considered precise enough, given the overall uncertainties involved in the ILUC factor calculations.

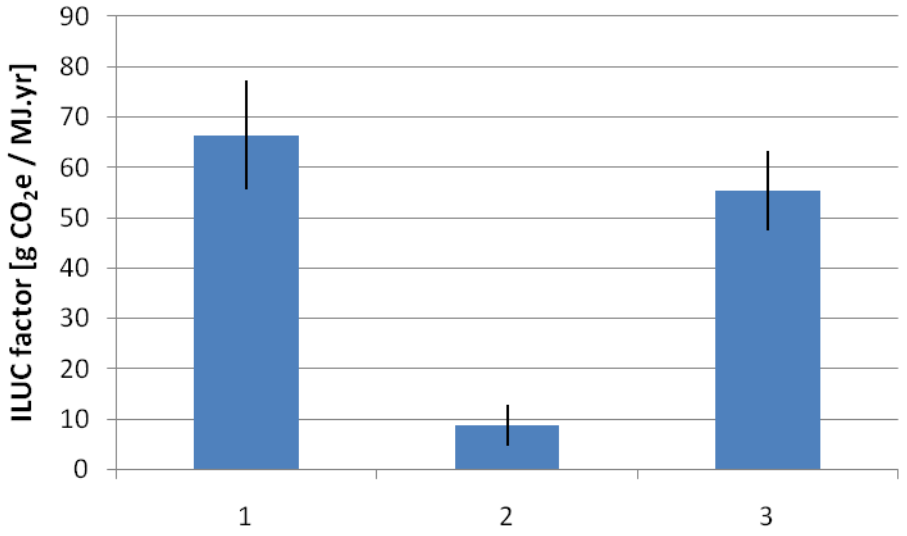

Figure 23. Indirect land use change impacts for the different scenarios modelled for soybean biodiesel.

As can be seen from Figure 23, the ILUC impact of soy biodiesel varies significantly when different assumptions are made about the type and source of oil used to replace it in the markets from where it is diverted. To summarise these effects:

- Palm oil is assumed to substitute the majority of the soybean oil that is diverted out of domestic markets. The soy ILUC impacts are therefore heavily dependent on the assumptions made about where and how that palm oil is grown. Scenarios 1 and 3 assume that the ILUC impacts of the palm oil are large; that all additional palm oil is produced through area expansion onto a large proportion of forest and peatland and that the palm plantation would be abandoned after one planting. Scenario 2 assumes that the ILUC impacts of palm oil are small; that only a small proportion of palm expands onto forest land and none onto peatland, and that palm will be grown for successive plantings on the land it expands onto.
- The ratio of substitution of soy oil by palm oil and rapeseed oil also affects the ILUC impacts. In a situation in which a greater proportion of rapeseed oil substitutes soybean oil, the ILUC impacts are smaller (contrast scenario 1, in which soybean oil in China is substituted by 50% rapeseed oil and 50% palm oil, and scenario 3, in which soybean oil in China is substituted by 90% rapeseed oil and 10% palm oil). However, the ILUC factor in scenario 3 is still large, as palm oil is still substituting a large proportion of soy oil in other non-Chinese markets: in scenario 1, 7.7 million tonnes palm oil substitute soy oil, compared with 2.6 million tonne rapeseed oil, and in scenario 2, 6 million tonne palm oil substitute soy compared with 4.3 million tonne OSR oil.
- The large confidence intervals on the bars represent estimates of the uncertainty associated with the carbon stock changes on the land and are calculated from Winrock analysis (US EPA, 2010)[18]. The magnitude of the uncertainty and the wide range in scenarios that could be explored again illustrate the difficulty in estimating the ILUC impacts of using a MJ of soybean biodiesel in 2020.

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[18] Apart from the data points for emissions from peatland and the carbon stock on the land for successive plantings of oil palm (see section 4.4.2).

Figure 24 illustrates the contribution of the different market responses to the final ILUC factor. Scenario 3 is shown as an example (the breakdown of the ILUC factors for the other scenarios are shown in Annex 5). It represents the case in which historical rates of forest conversion continue, 33% palm expansion is on peatland in Indonesia and Malaysia, the palm is only assumed to be grown on the land for one planting (and therefore the emissions are amortised over 30 years) and palm biofuel does not induce above baseline yield increases. The ILUC debits (i.e. carbon stock losses) are shown in red and the final ILUC factor for this scenario is shown in black.

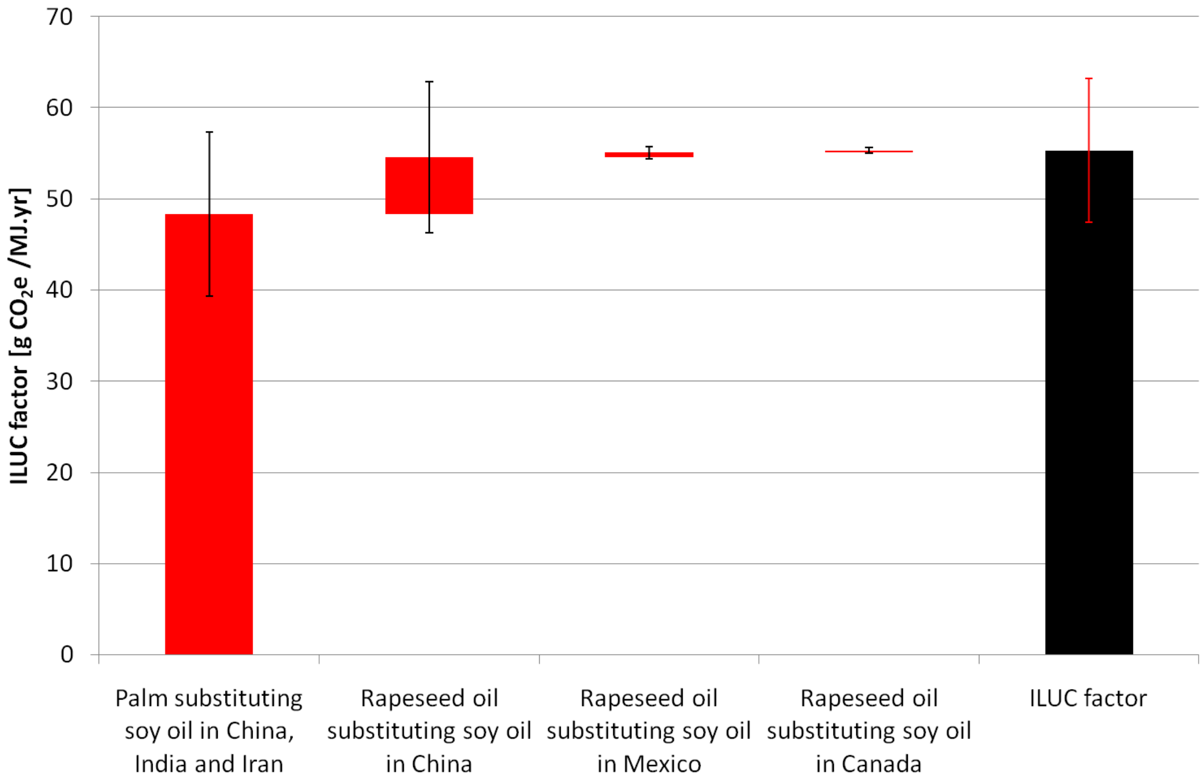

Figure 24. Waterfall diagram showing the contribution of each substitution to the overall ILUC factor for Soy biodiesel scenario 3.

This waterfall diagram highlights the large contribution of the palm ILUC factor to the overall ILUC impacts for soybean biodiesel and the key countries whose domestic vegetable oils consumption markets are assumed to be affected by an increased demand for soybean biodiesel.

Through exploring the different scenarios, the key message is that it is not possible to isolate the land use impacts of these different vegetable oils, as they strongly substitute for one another depending on the market location and need. As such, it is unlikely to be possible to identify “indicators” for low ILUC soybean practices in Argentina or Brazil, because the majority of these practices would have to relate to the management of palm in SE Asia, or OSR in Canada.

Although not strictly a means of reducing “indirect” land use change impacts, improving soybean yields, through means other than increased energy and carbon intensive inputs, will also help slow the rate of soybean expansion onto new land (and thereby reduce emissions). Many of these practices are already undertaken in key soybean growing areas.

7 Wheat bioethanol

7.1 Introduction

Wheat is, and is projected to remain, the main feedstock for bioethanol production in Europe. Wheat is a traditional crop in Europe with high yields. In fact, increases in yield have been enough to cover the increasing demand, and thus cultivation areas have been slowly decreasing over time.

Wheat is mainly used in Europe for food (bread, pasta) and animal feed, with each of these applications using a preferred type of wheat. In recent years, bioethanol production from wheat has grown. In order to simplify our modelling, we have looked at average data on all types of wheat for calculating the indirect land use change impact of wheat bioethanol.

Scenarios. Several scenarios have been considered for calculating the indirect land use impact of the additional demand for bioethanol from wheat in Europe in 2020. These scenarios explore different options for important model parameters, such as:

- the changes to the European trade balance of wheat, which will influence the geographical location of land use change;
- the above baseline yield increase for wheat which will influence wheat area expansion;
- the deforestation rates in different world regions. Although not directly linked to wheat, these rates influence the credit given to wheat for the production of the bioethanol co-product wheat DDGS;
- the possible uses of wheat bioethanol co-products (especially wheat DDGS).

Table 31 below shows how the parameters have been varied between the different scenarios. The exact numerical changes to the model for each scenario are discussed in the following sections.

Table 31. Overview of scenarios for the wheat to bioethanol chain.

Scenario / Parameter	1	2	3	4	5	6	7	8
Changes in European wheat trade balance	Yes (less exports)	Yes (more imports)	No	No	No	No	No	No
Yield increases due to biofuel demand	High	High	Low	High	High	High	High	High
Deforestation rates in Indonesia and Malaysia	Historical	Historical	Historical	Historical	Historical	Historical	Historical	10%
Deforestation rates in Argentina and Brazil	Historical	Historical	Historical	Historical	Historical	Resp. 24% and 38%	0%	Historical
Share of wheat DDGS used as animal fodder	100%	100%	100%	100%	50%	100%	100%	100%

This chapter starts with a short description of the baseline and biofuel projections for wheat. The possible market responses to the additional demand for wheat bioethanol are then discussed in section 7.3. Section 7.4 analyses the land use impact of the market responses and their GHG consequences, and Section 7.5 presents our results.

7.2 Additional demand for wheat bioethanol in Europe in 2020

As shown in section 3.2, the production of wheat in 2008 in Europe (including the small amount of wheat used for bioethanol production) is 151 million tonnes, based on data from FAPRI (2009a). The projection for wheat production in 2020 in the baseline was also based on FAPRI (2009a), but the amount of wheat used for bioethanol was kept constant at the 2008 level. The baseline production of wheat in 2020 is thus 148 million tonnes. From the assumptions described in section 3.2, the additional demand for bioethanol in 2020 in Europe will be 15 billion litres. Wheat is assumed to provide 67% of this bioethanol demand in 2020, resulting in a demand for wheat in 2020 in the biofuel projection of 26 million tonnes. However, when dealing with overall demand for wheat, the feed wheat displaced in Europe by biofuel co-products should be taken into account. Feed wheat displacement represents about 5 million tonnes of wheat (see section 4.3.5 for our assumptions on biofuel co-product treatment). Thus the demand for wheat in 2020 in the biofuel projection will be 170 million tonnes.

7.3 Market responses

In Europe, the additional demand for wheat could be met in a number of different ways:

1. Increased supply of wheat in Europe, either through substitution of wheat in the food or feed market by other cereals or by increased production of wheat in Europe;
2. Changes to the European trade balance of wheat, either through increased imports or lower exports;
3. Increased availability through improvements in efficiency in the supply chain;
4. Reduction in amount of wheat demand in other markets.

The other market response to be considered is the effect that co-products from wheat bioethanol production have on the markets in which they act as substituting products.

These market responses to the increased demand for wheat for bioethanol are mapped out in Figure 25. The magnitude of the effect of the different market responses is discussed further in this section and illustrated through the scenarios.

As discussed earlier in this report, market responses 3 and 4 have not been considered in this analysis. The extent to which these market effects make a contribution could be explored in future analysis of this type.

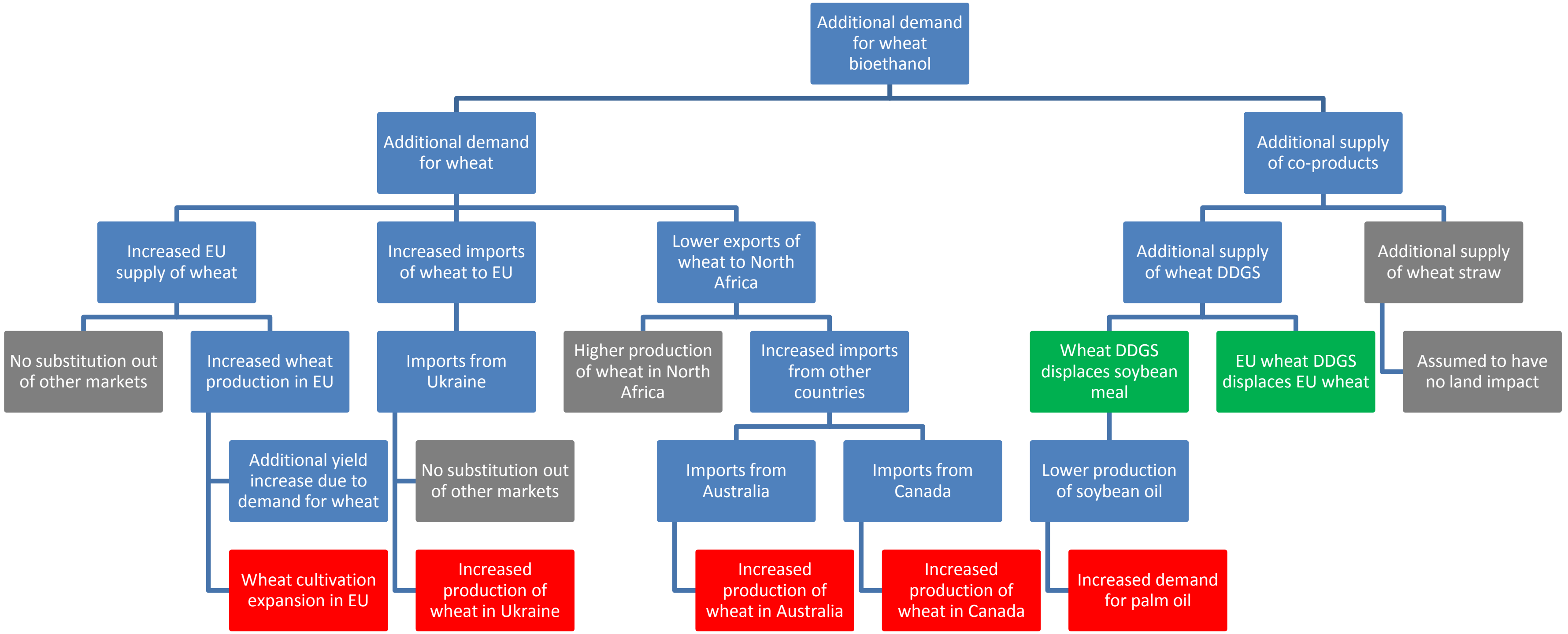

Figure 25. Market responses to an increase in demand for wheat bioethanol.

Land expansion, or ILUC "debits" are shaded red and avoided land expansion, or ILUC "credits" are shaded green. Market responses depicted in the diagram in grey are not considered further for reasons explained in the text.

7.3.1 Increased supply of European wheat

7.3.1.1 No displacement of wheat out of other markets

Wheat is currently grown for several purposes in Europe such as food (bread, pasta) or animal fodder. We assumed in this study that wheat would not be diverted out of the food market in Europe, due to its importance in European food and to the lack of possible substituting products.

However, the animal feed market is highly competitive and evolving. Figure 26 presents the evolution of the market share of most animal feed product in Europe, based on consumption data from USDA FAS (2010). From this graph, it is clear that the market share of wheat is not decreasing. Its market share has stayed relatively constant in recent years at around 35%, even with the increasing use of wheat for bioethanol production.

We have therefore also assumed that no substitution of wheat in the animal feed market would take place.

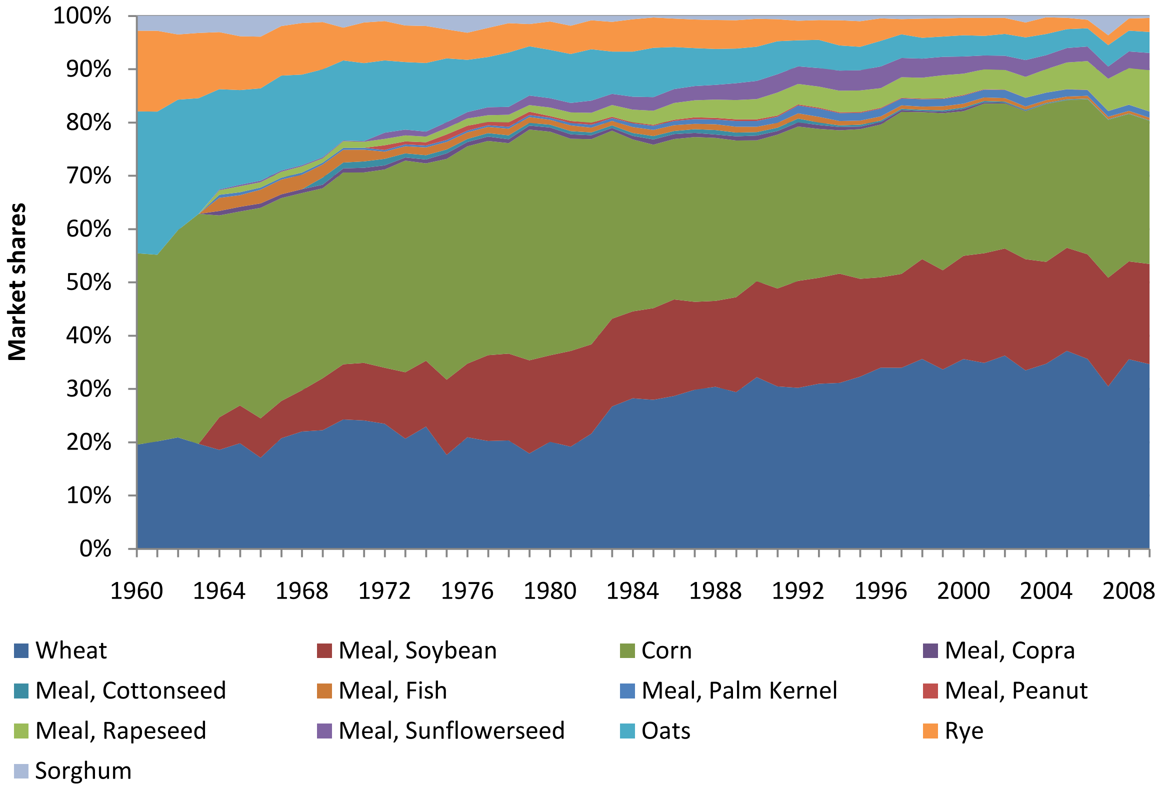

Figure 26. Historical trend in market shares of different animal feed in Europe from 1960 to 2009.

Source: USDA FAS (2010).

7.3.1.2 Increased European production of wheat

Europe has the potential to grow enough wheat to continue current levels of wheat export and meet the additional demand for wheat for bioethanol. However, it is considered by some stakeholders that a more likely response will be that the trade balance in Europe will change, e.g. the EU exports less wheat as a consequence of increased bioethanol production. In this sub-section, we look at the production of European wheat under different assumptions. In the next sub-section, we look at changes to wheat production outside of Europe due to changes in the European trade balance for wheat.

Yield and area change in Europe. To project the changes in yield and wheat cultivation area for the baseline and biofuels projection, the Lywood et al. (2009a) methodology was applied (see the methodology description in section 3.3.2).

In the baseline, the area of wheat grown is expected to decline between 2008 and 2020 due to increasing yields. It has been argued that such a big amount of land would not be left abandoned in the EU, but would continue to produce wheat for export. Such a scenario has not been considered explicitly in this section; however, it can be seen as a similar situation to scenario 1 which considers a decrease in European exports to meet the additional demand for wheat for bioethanol production. Indeed, if we had considered a scenario where the baseline production and export of wheat went up because of increasing yields and stable cultivation area, the additional European demand for wheat in the biofuel projection would come for one part from a decrease in exports, i.e. scenario 1. As scenario 1 has higher ILUC impact relative to the other scenarios – see Figure 29 page 96), the ILUC impacts of a scenario with high European exports in the baseline will also be in the high end of the range.

In the biofuel projection, the yield is expected to increase further while the decline in cultivation area is slowing down. The demand for European wheat depends on the scenarios considered. Scenario 1 and 2 look at how changes in European exports and imports of wheat would influence the ILUC factor. Thus in these scenarios, the above baseline demand for European wheat for bioethanol in 2020 will be lower than in all the other scenarios (leading to a lower yield increase and a lower cultivation area increase).

Table 32. Overview of production area and yield projections for wheat in Europe in the baseline and biofuel projections in 2008 and 2020.

Scenario:	All		Scenario 1		Scenario 2		Scenario 3		Scenario 4 to 8	
Projection:	**Baseline**		**Biofuels**		**Biofuels**		**Biofuels**		**Biofuels**	
Year:	**2008**	**2020**	**2020**	**Extra demand for biofuels**	**2020**	**Extra demand for biofuels**	**2020**	**Extra demand for biofuels**	**2020**	**Extra demand for biofuels**
Production [million t]	151	148	166	17.9	168	19.8	170	21.7	170	21.7
Area [million ha]	26.8	24.9	25.6	0.66	25.6	0.73	27.1	2.2	25.7	0.79
Yield [t / ha]	5.62	5.94	6.49	-	6.55	-	6.27	-	6.60	-

Yield change. The change in yield from 2008 to 2020 in the biofuels projection (from 5.62 to a maximum of 6.60 t/ha) corresponds to a maximum yearly increase of less than one percent, compared to an annual compound growth rate of slightly over one percent for the last 12 years period in Europe. We did not see any barrier to such an increase and the projected yields were thus assumed feasible.

However, to investigate the impact of attributing yield increases to additional biofuel demands, wheat ILUC scenario 3 looks into what would happen if the yield compound annual growth rate was limited to half of that calculated earlier. The yield would only increase to 6.27 t/ha in 2020 in the biofuel projection. This would increase the wheat cultivation area needed to cover the total additional demand for wheat to 27.1 million hectares, about 302’000 ha more than today’s wheat cultivation area.

A number of stakeholders expressed concern that these yield improvements would be met in part through an increase in fertiliser application, which would consequently have GHG emissions associated with its production and use. However, these emissions have not been included in our calculations here, but we have provided a discussion of this issue in section 3.3.4.2 and our rationale for not including these emissions in these calculations. That section concludes that the magnitude of this effect on the ILUC factor is unlikely to be large, relative to the other ILUC effects being considered.

Land availability in Europe. For all scenarios except scenario 3, the additional demand for wheat for bioethanol production results in a maximum increase in wheat cultivation area of 0.8 million hectares. However the total wheat cultivation area in 2020 in the biofuel projection is still lower than the current wheat cultivation area. Such an expansion is thus assumed to be feasible, and that this total additional demand for wheat could be produced in Europe.

For scenario 3, the wheat cultivation area in 2020 in the biofuel projection will exceed the current wheat cultivation area by 302,000 ha. We have assumed that enough suitable land would be available in Europe in 2020 to cover such an increase.

7.3.1.3 Change in the wheat trade balance

In most scenarios, it was considered that the additional wheat demand for bioethanol production will be produced domestically. However, two scenarios were built around changes in the wheat trade balance to test the effect of this hypothesis on the wheat ILUC factor.

Increase in imports. In scenario 2 (Table 31), an above baseline increase in imports was assumed. Currently, the main exporter of wheat to the European Union is the Ukraine.

Our projections of Ukrainian wheat exports to Europe by 2020, for both the baseline and the biofuel projection, are based on data from FAOSTAT (2010b). Data on Ukrainian exports are available from 2002 to 2005. The compound annual growth rate of Ukrainian wheat exports to Europe in this period was calculated to be 4%.

In our baseline, we assumed that this trend would continue until 2020, leading to an import of 3.8 million tonnes of wheat by 2020 from the Ukraine. In our biofuels projection, for scenario 2, we assumed that this import would increase by 50% (i.e. an additional import of 1.9 million tonnes) which would be used in Europe for bioethanol production. These amounts seem reasonable in light of the 25 million tonnes of wheat produced by Ukraine in 2008 (FAPRI, 2009a).

Lower exports. In scenario 1 (Table 31), a decrease in European exports of wheat was assumed. Currently, Europe’s main wheat importer is North Africa (mainly Algeria, Morocco and Egypt[19]). Scenario 1 assumes a reduction of 50% in the exports of Europe to that region.

Data on North African wheat imports were taken from FAOSTAT (2010b). Figure 27 presents the historical trends in total imports of Algeria, Egypt and Morocco. These trends have been extrapolated

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[19] In the following sections, we have taken Algeria Morocco and Egypt to represent North Africa.

to estimate the total imports of these countries in 2020. We have also assumed that the European share in these imports by 2020 will be the same as in 2005 for the baseline (respectively 54%, 19% and 29% for Algeria, Egypt and Morocco). Through this approach the EU exports to North Africa in 2020 in the baseline amount to 7.5 million tonnes of wheat. This is almost the same results as if FAPRI's projections on net imports of Algeria, Egypt and Morocco to 2018 were extrapolated to 2020 (which gives an increase of 7.4 million tonnes of wheat imports).

For the biofuel projection, in scenario 1, we have assumed that European wheat exports to North Africa will be 50% lower than in the baseline (i.e. a decrease of 3.8 million tonnes of wheat).

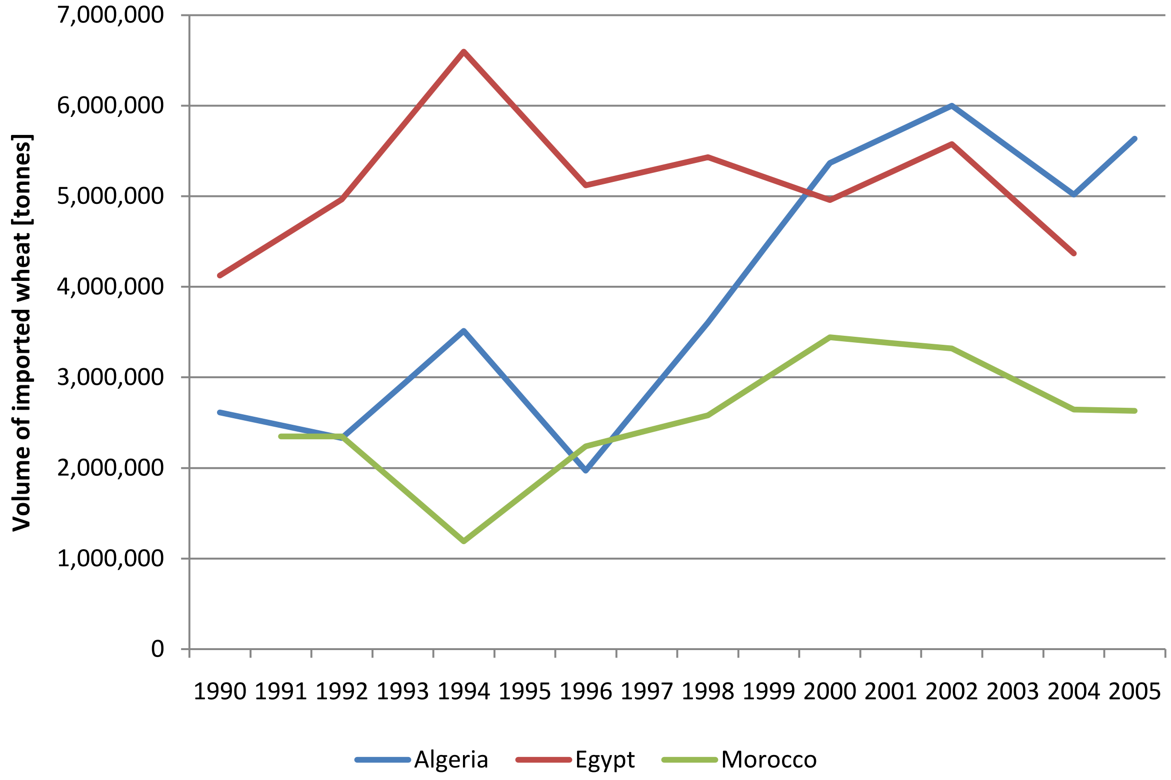

Figure 27. Historical trend in wheat imports for Algeria, Egypt and Morocco.

Source: FAOSTAT (2010b).

7.3.2 Increased supply of non-European wheat

As determined in the previous sub-sections, changes in European wheat trade will have an impact on Ukrainian exports and North African imports. In the following sub-sections, the additional supply of wheat in these two regions is analysed.

7.3.2.1 Increased exports from Ukraine to Europe

Increased Ukrainian production of wheat. We have assumed the increased exports from the Ukraine will come from an increase in production of wheat within that country. As in Europe, wheat area in the Ukraine is set to decline in the baseline. The additional demand due to European demand for biofuels will reduce the rate of decline in the harvested wheat area in Ukraine, but will not change the trend, as can be seen in Table 33.

Table 33. Overview of production area and yield projections for wheat in Ukraine in the baseline and biofuel projections in 2008 and 2020.

Projection:	Baseline		Biofuel	Additional demand for biofuels
Year:	**2008**	**2020**	**2020**	**2020**
Production ['000 tonnes]	25,500	22,146	24,033	1,888
Area ['000 ha]	7,000	6,311	6,444	132.6
Yield [tonne / ha]	3.64	3.51	3.73	-

Yield improvements and land availability in Ukraine. Table 33 shows the results of the Lywood et al. (2009a) methodology for wheat production in the Ukraine (methodology described in section 3.3.2.2). In the baseline, wheat yield actually decreases between 2008 and 2020. In the biofuel projection, the yield increases slightly between 2008 and 2020.

The increase in cultivation area that is estimated to be required for the additional demand for wheat biofuel (i.e. 132,600 ha) is assumed to be feasible, as it only reduces the decline in cultivated area. Furthermore, as discussed in section 5.3.2.1, there are large amounts of abandoned agricultural lands in Ukraine that make a further agricultural area expansion feasible.

7.3.2.2 Lower exports to North Africa

Lower wheat export from Europe to North Africa means that North African countries would have to find another wheat source, which can either be an increase in their domestic production or an increase in imports of wheat from non-European countries.

Production of wheat in North Africa. Egypt has seen its production of wheat gradually increasing between 1990 and 2005. However, wheat production in Algeria and Morocco has been much more variable (see Figure 28). We have thus assumed that North Africa would not be able to scale up its production of wheat to compensate for the lower exports from Europe. It is more likely that North Africa will increase its imports of wheat from other regions, which we have assumed to be Australia and Canada as these were the two major wheat exporters to the region in 2005, after the European Union (FAOSTAT, 2010b). We assumed each of these countries would provide 50% of the decrease EU exports.

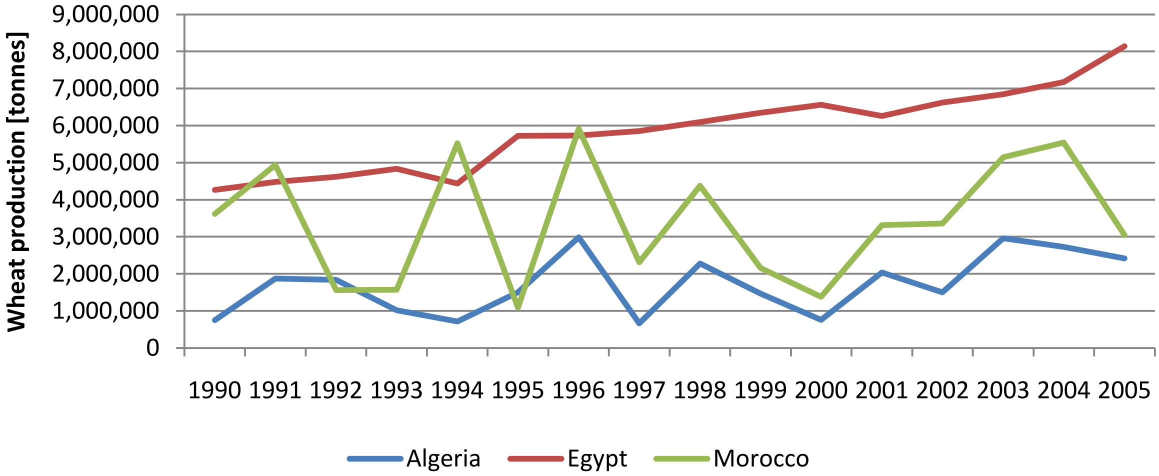

Figure 28. Historical trend in wheat production in Algeria, Egypt and Morocco.

Source: FAOSTAT (2010b).

Increased North African imports of wheat from Australia. The amount of land required in Australia to produce the additional demand for wheat was estimated based on projection in yields by FAPRI (2009a). FAPRI projects wheat yield in Australia to be 1.92 tonnes/ha in 2018. The projection trend can be extrapolated to 2020, to give a yield of 2.00 tonnes/ha, which would mean the conversion of 0.9 million hectares of land into wheat cultivation.

Increased North African imports of wheat from Canada. The same approach was followed to estimate the amount of land required in Canada to produce the additional demand for wheat. Based on FAPRI (2009a), we estimated that yields in Canada in 2020 will be 2.87 tonnes / ha, corresponding to an area of 0.7 million hectares of land required for wheat production.

7.3.3 Effect of co-products

The production of bioethanol from wheat has one main co-product: wheat distiller's dried grains with solubles or DDGS. Furthermore, the cultivation of wheat also produces rapeseed straw. However, for this study we have considered that wheat straw does not displace any land-grown products. We assumed that wheat DDGS was more likely to have a bigger impact on the ILUC factor than wheat straw, and thus concentrated our co-product analysis on the DDGS. This assumption could be explored in more detail in future analysis of this type.

Thus the only co-product considered to have land impacts is wheat DDGS. Wheat DDGS is used as an animal feed (Lywood et al., 2009b; Schmidt and Weidema, 2008) and as such replaces soybean meal and feed wheat. The approach taken to estimate the land impact of soybean meal and feed wheat substitution is described in section 4.3.5.

In assessing the impact of wheat DDGS, two different possibilities were considered based on the assumption about the share of wheat DDGS that would actually be used for animal feed. Actually, the domestic production of protein feed cannot cover the European demand, which leads to significant imports of meals such as soybean meal. As such, wheat DDGS would be expected to have a reasonable market. However, as more protein meals become available due to biofuel production, there may be less of an incentive for wheat DDGS to be used.

To determine the ratio of wheat DDGS that will be used for different purposes, it would have been necessary to model the animal feed market, taking into account questions such as the size of the shortfall in protein feed, the amount of DDGS that could be taken up before reaching saturation, whether there would be an export market for the surplus European DDGS. We have not conducted such a model. However, to explore sensitivity to this issue, we examined the effect of only using 50% of the DDGS in scenario 5 compared with 100% of wheat DDGS used in all the other scenarios (see Table 31 for a summary of the scenarios).

7.4 Land use impacts and greenhouse gas consequences

Land impacts were calculated based on the difference in harvested area between the baseline and the biofuel projection in 2020 using the land use type and associated GHG emissions analysis carried out by Winrock International for the U.S. EPA and applied in the Renewable Fuel Standard 2010 (RFS 2). Winrock's approach is described in further details in section 4.4.1 (U.S. EPA, 2010).

In the following sections we have presented the type of land use impact of the production of bioethanol from wheat in each of the producing regions and the related GHG consequences. Finally, the land impacts of co-products are presented in the last section.

7.4.1 Land use change from European supply

The amount of land that is likely to be taken up by wheat production in Europe is determined in Section 7.3.1.2. For all scenarios except scenarios 1 to 3, wheat cultivation area is expected to expand by 788,000 hectares. For scenario 1 and 2, this area expansion is lowered to 662,000 and 726,000 hectares respectively. For scenario 3, the area expansion is 2.2 million hectares.

Type of land expanded onto. Wheat cultivation areas are decreasing in both the baseline and the biofuel projection (except in scenario 3). The total decrease in area in the baseline for wheat is 1.9 million hectares, and thus covers the 788,000 hectares of maximum expansion (excluding scenario 3) that would be caused by additional demand for wheat for biofuels. However in the baseline, this 0.8 million hectares would not have been grown with wheat. As the land used for wheat cultivation is usually of high quality, it is unlikely that the land would have been abandoned. It is more probable that some other crops (barley or rye for example) would have been grown on the land. However, this does not mean that barley or rye cultivation area would have increased but rather that it would have shifted location – leading to other land being freed that would then have been used to grow something else, etc. This knock-on effect ultimately ends with some land being abandoned. Thus the land use difference between the baseline and the biofuel projection due to increased wheat demand in Europe is land that would have been abandoned in the baseline but that is used for crop production in the biofuel projection. Furthermore, as average yields are used[20], the amount land that would have been abandoned is equal to the amount of wheat expansion in the biofuel projection, i.e. 788,000 hectares.

The otherwise abandoned land would have accumulated some carbon stocks, the amount of which depends on the type of land it would have become. Table 20 on page 61 shows the share of the

[20] See section 3.3.2.2 for a discussion on why we have used average yields.

different types of land considered with the carbon stock this land would have accumulated over 30 years (i.e. reversion factors).

For scenario 3, the wheat cultivation area is actually increasing in the biofuel projection, from 26.8 million hectares in 2008 to 27.1 million hectares in 2020. This means that part of the additional 2.2 million hectares of land required for the biofuel feedstock cultivation will be land that would have ultimately been taken out of cultivation in the baseline (1.9 million hectares) and the remaining part is actual wheat cultivation area expansion onto land not under arable production (302,000 hectares). Table 34 presents the types of land crop and pasture (and therefore wheat) may expand onto and the GHG consequences of such an expansion.

Table 34. Types of land use change and emission factors associated with them for the European Union. Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year reversion factors	95% confidence interval
Forest	6%	107 t CO_2e / ha	± 90 t CO_2e / ha
Grassland	25%	20.9 t CO_2e / ha	± 24 t CO_2e / ha
Mixed	32%	39.4 t CO_2e / ha	± 26 t CO_2e / ha
Savannah	21%	25.5 t CO_2e / ha	± 23 t CO_2e / ha
Shrub land	14%	34.6 t CO_2e / ha	± 23 t CO_2e / ha
Wetland	1%	27.8 t CO_2e / ha	± 23 t CO_2e / ha
Barren	1%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

7.4.2 Land use change from Ukrainian supply

In scenario 2, import of wheat from the Ukraine is assumed. The land use impact of such an increase in production was determined in section 7.3.2.1.

Type of land expanded onto. Just as for Europe, wheat cultivation area in Ukraine is decreasing in the baseline. In the biofuel projection, for scenario 2, the additional amount of wheat production in the Ukraine is not enough to reverse this trend. Thus reversion data from Winrock International (US EPA, 2010) was also used to determine the share and carbon stock of the type of land use impact this additional wheat production.

Table 35 presents the type of land that would have replaced abandoned cropland/pasture in Ukraine and the foregone emissions associated with this conversion no longer taking place in the biofuel projection.

Table 35. Types of land use change and emission factors associated with them for Ukraine. Based on Winrock data for the US EPA (2010).

Type of land	Share	Reversion data	
		30 year reversion factors	95% confidence interval
Forest	34%	68.2 t CO_2e / ha	± 5.9 t CO_2e / ha
Grassland	14%	14.5 t CO_2e / ha	± 17 t CO_2e / ha
Mixed	32%	35.0 t CO_2e / ha	± 10 t CO_2e / ha
Savannah	16%	18.5 t CO_2e / ha	± 10 t CO_2e / ha
Shrub land	5%	26.5 t CO_2e / ha	± 11 t CO_2e / ha

7.4.3 Land use change from Australian and Canadian supply

Scenario 1 considers that wheat cultivation will expand in Australia and Canada, due to the Australian and Canadian wheat exports increasing to replace European exports to North Africa. Based on projections by FAPRI (2009a), harvested area of wheat is increasing in Australia, so an additional demand for wheat is likely to accentuate this trend.

However, in Canada, the total harvested area[21] of wheat is projected to decrease. Using the compound annual growth rate of the harvested area of wheat between 2008 and 2018, we projected the area for 2020. Between 2008 and 2020, we expect a reduction of about 0.7 million hectares, equal to the amount of land needed for additional wheat cultivation due to European demand for bioethanol (see Section 7.3.2.2). We have assumed that this land would have been left idle in the baseline, meaning that the additional demand for wheat leads to avoided land reversion back to "natural" land.

Table 36 presents the type of land that could be converted to cropland, together with the GHG emissions due to such a land conversion for Australia and Table 37 shows reversion data for Canada.

Table 36. Types of land use change and emission factors associated with them for Australia. Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year reversion factors	95% confidence interval
Forest	6%	127 t CO_2e / ha	± 73 t CO_2e / ha
Grassland	32%	13.1 t CO_2e / ha	± 20 t CO_2e / ha
Mixed	11%	40.8 t CO_2e / ha	± 26 t CO_2e / ha
Savannah	22%	16.7 t CO_2e / ha	± 18 t CO_2e / ha
Shrub land	25%	23.9 t CO_2e / ha	± 18 t CO_2e / ha
Wetland	0%	18.5 t CO_2e / ha	± 17 t CO_2e / ha
Barren	4%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

[21] The total Canadian harvested area was calculated based on the FAPRI (2010) projection in harvested area for the following crops in Canada: barley, corn, wheat (all), rapeseed, soybean and sugar beet.

Table 37. Types of land use change and emission factors associated with them for Canada. Based on Winrock data for the US EPA (2010).

Type of land	Share	Reversion data	
		30 year reversion factors	95% confidence interval
Forest	43%	42.1 t CO_2e / ha	± 15.8 t CO_2e / ha
Grassland	11%	16.2 t CO_2e / ha	± 142 t CO_2e / ha
Mixed	4%	31.7 t CO_2e / ha	± 16.3 t CO_2e / ha
Savannah	15%	19.1 t CO_2e / ha	± 16.6 t CO_2e / ha
Shrub land	26%	25.1 t CO_2e / ha	± 16.0 t CO_2e / ha

7.4.4 Land use change due to co-products

Co-products of the wheat to bioethanol chain were discussed in section 7.3.3, where the scenarios on the uses of co-product use were also introduced.

7.4.4.1 Land use impacts and associated greenhouse gas consequences of soybean meal displacement

Avoided land use change in Argentina and Brazil. The displacement of soybean meal means that less soybean will be grown. Major soybean producers are Argentina and Brazil and it is expected that they will see a decrease in demand for their soybean meal and thus for soybeans. The wheat DDGS then earns an ILUC factor "credit" to the wheat bioethanol because less demand for soybean means reduced soybean cultivation area expansion and so avoided land use change.

Table 23 on page 64 presents the type of land soybean cultivation would have expanded onto and the GHG emissions saved through avoiding this land use change, for both Argentina and Brazil.

To test the impact forest protection policies in Argentina and Brazil could have on ILUC factors of European demand for wheat bioethanol, we considered two scenarios:

- Scenario 6 considers a higher rate of deforestation in Argentina and Brazil. This also meant lowering the rate of conversion of grassland, mixed land and savannah to cropland.
- Scenario 7 considers a lower rate of deforestation in Argentina and Brazil (i.e. a good forest production policy). This also meant increasing the rate of conversion of grassland, mixed land and savannah to cropland.

Table 38 below shows the types of land we have considered to be converted into cropland for these two scenarios (based on the Winrock data for land typically converted to cropland/pasture land).

Table 38. Share of land types converted to cropland in Argentina and Brazil for scenarios 6 and 7. Based on Winrock data for the US EPA (2010).

Land type	Scenarios 1-5 & 8 (historical deforestation)		Scenario 6 (high deforestation)		Scenario 7 (low deforestation)	
	Argentina	Brazil	Argentina	Brazil	Argentina	Brazil
Forest	12%	19%	24%	38%	0.0%	0.0%
Grassland	26%	18%	20%	12%	32%	25%
Mixed	27%	20%	21%	14%	33%	27%
Savannah	17%	35%	17%	29%	17%	42%
Shrub land	14%	6%	14%	6%	14%	6%
Wetland	1%	0.14%	1.0%	0.14%	1.0%	0.14%
Barren	3%	0.33%	3.0%	0.33%	3.0%	0.33%

Consequences of displaced soybean meal on vegetable oils. The decrease in soybean production means that less soybean oil (co-product of the soybean meal production) will be produced. This soybean oil will have to be replaced by other vegetable oils. We have considered that palm oil will be replacing soybean oil as it is the current marginal oil and is expected to remain so until 2020.

To account for the indirect land use change impacts of this increase in palm oil demand, we have used the ILUC factors for palm as calculated in section 4 of this report. Table 39 below summarises which palm oil scenarios were used in each of the wheat scenarios.

Table 39. Matching palm and wheat scenarios.

Wheat scenario	1	2	3	4	5	6	7	8
Palm scenario	3	3	3	3	3	3	3	6

Most wheat scenarios were associated with the palm scenario 3. This palm scenario was selected based on its parameters: historical deforestation rates, 5% of palm expansion onto peatland and use of 30 years emission factors.

Wheat scenario 8 corresponds to testing the influence of forest protection policies in Indonesia and Malaysia on the ILUC factor for wheat bioethanol. This was done by lowering the deforestation rate in Indonesia and Malaysia from the historical trend to 10%. Of course this was linked to higher conversion rates of other land types. The palm scenario that best represented this assumption is palm scenario 6.

7.4.4.2 Land use impacts and associated greenhouse gas consequences of feed wheat displacement

The displacement of European-grown feed wheat by European wheat DDGS (i.e. DDGS produced in Europe) leads to a lower demand for wheat in Europe. A lower wheat production in Europe leads to more agricultural land being abandoned and thus reverting to high carbon stocks. Thus the credit for displacing feed wheat is calculated using the reversion data from Winrock International (US EPA, 2010). Table 20 on page 61 shows this data for Europe.

7.4.4.3 Greenhouse gas consequences of wheat DDGS not used as animal fodder

In scenario 5 (Table 31), we have considered that only 50% of the produced wheat DDGS would be used as animal fodder. A likely other use for wheat DDGS is co-firing in power plants to produce electricity. We have not considered that the ILUC factor should receive a credit for this alternative use, as there would be no ILUC impacts associated with displacing coal, and the emissions saved from co-firing would be credited to the calculation of the “direct” GHG emissions.

This effect cannot be considered as an ILUC impact, but was included in this study to ensure consistency in the treatment of co-products, and to allow for the comparison of the GHG benefits of different co-product uses.

7.5 Scenario results

Figure 29 presents the ILUC factors for all eight wheat bioethanol scenarios. For all the scenarios, the ILUC factor is negative. This means that the GHG credit given to the wheat DDGS for avoiding some land use change is in all scenarios bigger than the GHG consequences of the actual land use change caused by the additional production of wheat. So the production of wheat DDGS more than compensates for the production of wheat – in terms of GHG emissions.

The variation between the different ILUC factors is large, compared for example to the variation in the calculated ILUC factors for oilseed rape biodiesel.

No central scenario is provided, as it was felt that it would be too difficult to assign probabilities to each of these scenarios.

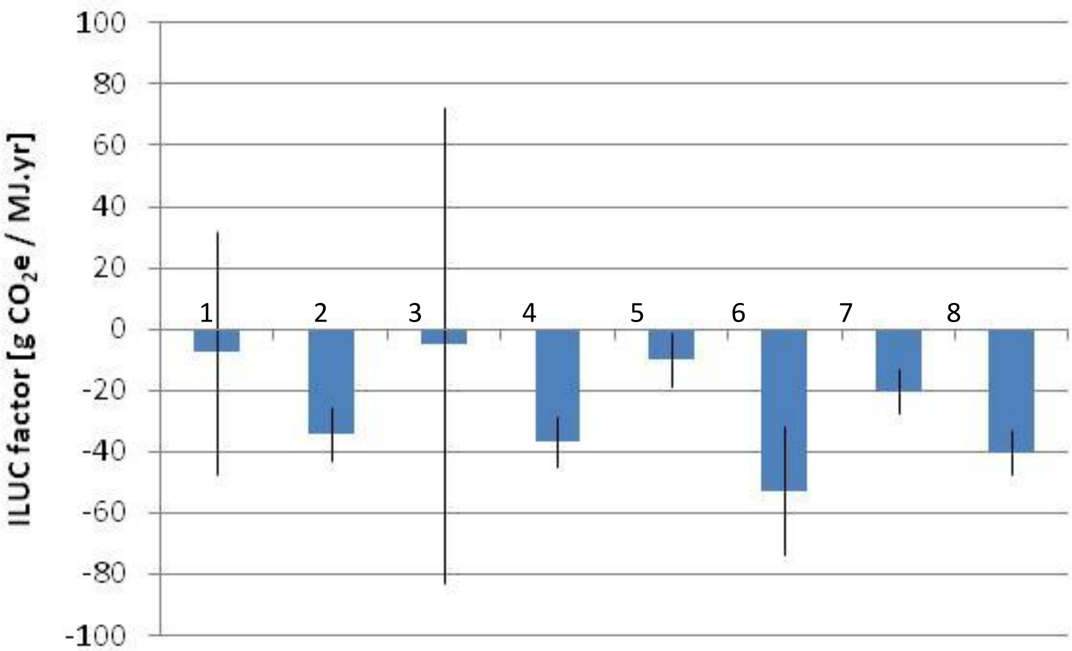

Figure 29. Indirect land use change impacts for the different scenarios modelled for wheat bioethanol.

Based on the analysis carried out for these scenarios, parameters that increase the ILUC factor for wheat bioethanol can be identified:

- Scenario 5 shows the difference made if only 50% of the wheat DDGS is used as animal fodder. Given the large quantity of animal feed imported into the EU, it was considered that there would be a market for a significant proportion of the DDGS to be used as animal feed.
- Scenario 7 indicates that lower deforestation rates in Argentina and Brazil would increase the ILUC factor. This is counter-intuitive as lower deforestation rates means less GHG emissions globally. However, as deforestation rates in Argentina and Brazil are only taken into account in the wheat chain through co-product credits, lower deforestation rates would mean that wheat DDGS is avoiding less deforestation and thus gets a lower credit.
- Scenario 3 shows the impact of assuming a lower above-baseline yield increase in the biofuels projection. As expected, lower yields lead to significantly higher ILUC impacts (i.e. closer to zero in this case) as it means more land will have to be used for wheat cultivation for bioethanol.
- Changes to the European wheat trade balance have an important impact on the ILUC factor, especially decreasing exports. This is probably due to the fact that increases in imports are assumed to trigger higher production in Ukraine, with low GHG emission associated to the land use change, whereas decreasing exports are assumed to lead to higher production in Canada and Australia that have high emission factors for land use change.

Fewer tested parameters actually seem to decrease the ILUC factor. These are mostly linked to deforestation rates:

- Scenario 6 (which is the opposite of scenario 7 in terms of deforestation rates in Argentina and Brazil) actually decreases the ILUC factor (where scenario 7 increases it).
- A lower deforestation rate in Indonesia and Malaysia means that we use a lower ILUC factor to represent the ILUC impact of increased demand for palm oil due to lower soybean oil production due to soybean meal displacement by DDGS. Thus the overall wheat ILUC factor is reduced (scenario 8).

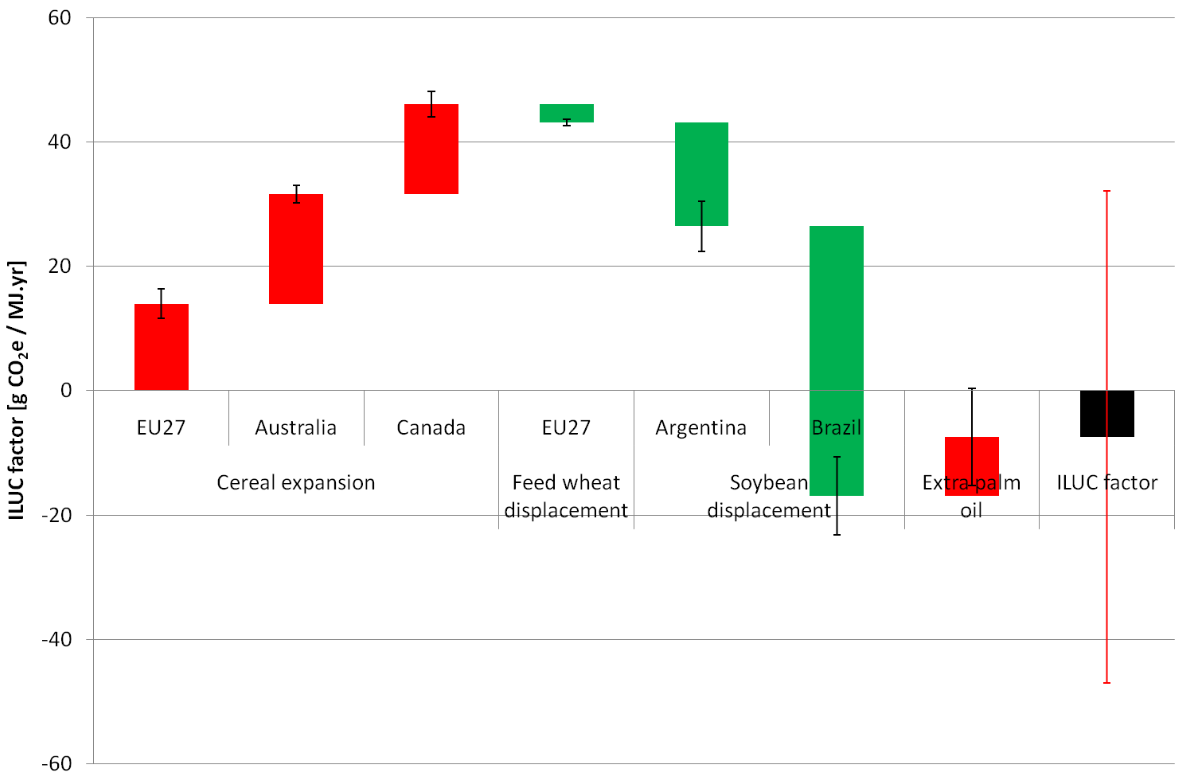

Figure 30. Waterfall diagram showing the contribution of each market response to the overall ILUC factor for scenario 1.

Figure 30 above shows the contribution of each market response to the overall ILUC factor. This is an example based on scenario 1. For the breakdown of the ILUC factor for the other scenarios, please refer to Annex 6.

It is clear from Figure 30 that the major contributor to the ILUC factor is the GHG credit due to the displacement of soybean meal by wheat DDGS, especially in Brazil. In this study, we considered that 1 t of wheat produces 0.29 t of bioethanol and 0.33 t of DDGS. The relatively higher importance of the wheat DDGS than the wheat production in terms of ILUC associated GHG emissions is explained by the fact that the wheat DDGS is avoiding the displacement of land with much higher carbon stock than the land the wheat production is expanding onto. Other key contributions are soybean meal displacement in Argentina and wheat production in the European Union, Australia and Canada.

Scenario 1 is thought to be one of the most realistic wheat bioethanol ILUC scenarios. It can be seen that the ILUC impact of scenario 1 is slightly negative (around -5 g CO_2e / MJ bioethanol). If one were to account for, for example, GHG emissions due to additional fertiliser application for yield increases, the ILUC impact could become positive (see discussion in section 3.3.4.2).

No “preferred” or “central” scenario has been put forward as the most likely for 2020, as a range of scenarios better reflects the remaining uncertainty around what will happen to 2020. It has also only been possible to show here a limited number of scenarios. However, one can refer to the data provided in the annexes for the different scenarios to understand how the ILUC factors would alter had more scenarios been carried out. For example, the ILUC factor for a scenario with increased wheat imports and 50% use of co-product as animal feed can be approximated using the data provided in Annex 6.

8 Bioethanol from sugarcane

8.1 Introduction

Sugarcane is currently the second largest feedstock for bioethanol production, after corn in the US. Sugarcane has been used since the 1970's in Brazil for bioethanol production, encouraged by the Government through the Proálcool program as a reaction to rising oil prices. More recently, Brazilian exports of sugarcane bioethanol have increased, and production in other countries is also starting.

The direct greenhouse gas savings of sugarcane bioethanol are high compared to other bioethanol routes, and have been recognised as such in policy. In Europe, sugarcane bioethanol meets the highest greenhouse gas saving threshold set by the Renewable Energy Directive, i.e. greater than 60% savings (European Commission, 2009a). In the USA, sugarcane bioethanol is currently the only commercially available bioethanol to be classified as an advanced biofuel, i.e. surpassing the 50% GHG emission saving threshold (US EPA, 2010b).

Scenarios. Several scenarios have been considered for calculating the indirect land use change impact of the additional global demand for bioethanol from sugarcane in 2020. These scenarios explore different possibilities for important model parameters, such as:

- different levels of demand for bioethanol from sugarcane in Europe and in the US, to reflect the uncertainty around the readiness of other (second generation) biofuel technologies in 2020 to deliver such high GHG emissions savings;
- variations in regions assumed to increase production of sugarcane in response to global sugarcane ethanol demand;
- the level of above baseline yield increase of sugarcane grown in different regions;
- the country in which crop land will ultimately expand onto natural land, as a result of sugarcane displacing crop land in Brazil;
- the link between increased demand for agricultural crops and pasture displacement in Brazil, i.e. whether or not additional demand for sugarcane is assumed to specifically cause pasture displacement ;
- the rate of pasture intensification in different regions in Brazil;
- the deforestation rate in Brazil.

Table 40 below shows how the parameters have been varied between the different scenarios. The exact numerical changes to the model for each scenario are discussed in the following sections.

Table 40. Overview of scenarios for the sugarcane to bioethanol chain.

Parameter \ Scenario	1	2	3	4	5	6	7	8	9	10	11	12	13	14
Low EU and US demand for SC bioethanol	✓	✓	✓		✓		✓	✓	✓			✓	✓	✓
High EU and US demand for SC bioethanol				✓		✓				✓	✓			
Small domestic production in US and small export from Southern Africa to EU	✓			✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
No domestic production in US and no export from Southern Africa		✓	✓											
Above baseline yield increase in all countries producing additional sugarcane	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	
No above baseline yield increase in all countries producing sugarcane														✓
Pasture displacement happens regardless of the demand for sugarcane	✓	✓	✓	✓	✓	✓								✓
Additional demand for sugarcane causes pasture displacement							✓	✓	✓	✓	✓	✓	✓	
Historical rate of pasture intensification in the Centre-South and North region	n/a	n/a	n/a	n/a	n/a	n/a	✓		✓	✓				n/a
Higher rate of pasture intensification in the North than in Centre-South region	n/a	n/a	n/a	n/a	n/a	n/a		✓			✓			n/a
Identical rate of pasture intensification in the Centre-South and North regions	n/a	n/a	n/a	n/a	n/a	n/a						✓		n/a
Lower rate of pasture intensification in North than in Centre-South region	n/a	n/a	n/a	n/a	n/a	n/a							✓	n/a
All crop cultivation area displaced to Brazil	✓	✓		✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Soy cultivation area displaced to Argentina, other crops displaced to Brazil			✓											
Historical deforestation rate in Brazil	✓	✓	✓	✓			✓	✓		✓	✓	✓	✓	✓
Lower deforestation rate in Brazil					✓	✓			✓					

This chapter starts with a short description of the baseline and biofuel projections for sugarcane. The possible market responses to the additional demand for sugarcane bioethanol are then discussed in section 8.3. Section 8.4 analyses the land use impact of the market responses and their GHG consequences, and section 8.5 presents our results.

8.2 Additional global demand for sugarcane bioethanol in 2020

Two different pathways are possible for sugarcane conversion to bioethanol:

- Sugarcane is crushed leading to the production of sugarcane juice. This juice can be converted to sugar and molasses. Molasses can be used to produce bioethanol through a classical fermentation process.
- Or the juice can be fermented directly to produce bioethanol.

In this study, we only consider the second pathway, i.e. the indirect land use change impact of increased demand for bioethanol from sugarcane juice. The reason for this is that it is assumed that sugar is the primary product and molasses the dependent co-product. As such, using molasses for bioethanol production is not assumed to affect the area grown for sugarcane. However, doing so may result in other effects, for example, if molasses is usually (in the baseline) converted to ethanol for use in the beverages industry, and is then diverted to transport fuel (in the biofuel projection), the beverage industry will need to be supplied with ethanol produced in some other way. The emissions associated with the production of this additional ethanol should be attributed to the use of molasses ethanol as biofuel. Thus, the indirect impacts of using molasses ethanol for transport fuel will be different from sugarcane juice ethanol, and as such they have not been included in this analysis.

The next section looks into the contribution molasses could make to meeting bioethanol demand. Section 8.2.2 then presents our two sugarcane ethanol scenarios.

8.2.1 Contribution of molasses to bioethanol production in 2020

Whereas in Brazil most of the bioethanol is produced from sugarcane juice, in the rest of the world, it is actually molasses bioethanol that dominates (Gopal and Kammen, 2009). For example, a typical Indian sugar mill produces on average 10 t of sugar and 4 t of molasses, from which ethanol is produced, per 100 t of sugarcane (SSI, 2009).

In section 3.2, we estimated the demand for sugarcane bioethanol to be 69 billion litres by 2020. Part of this demand will be fulfilled by bioethanol produced from molasses. The potential for molasses production in different world regions was estimated based on:

- total sugar production for food purposes in main countries of each region as projected by FAPRI (2010);
- a conversion factor of sugarcane to molasses of 0.035 t molasses / t sugarcane (average of the conversion factor reported by FAO, 1986; Foodmarketexchange.com, 2003; de Armos and Almazan, 2006; Sritoth, 2007 and Terajima et al., 2005);
- the assumption that 50% of all produced molasses could be used to produce fuel bioethanol[22] – this is based on the assumption that fuel use would not outcompete all other uses of molasses ethanol (e.g. the beverage industry).

By taking into account molasses bioethanol, the demand for sugarcane juice bioethanol was lowered to 64 billion litres by 2020. Table 41 below summarises these calculations. As can be seen in this table, bioethanol from molasses in 2020 is assumed to amount to ~6 billion litres. However, this could in theory vary from 0-12 billion litres if none or all of the molasses is used to produce bioethanol for transport fuel. Therefore, the demand for sugarcane juice bioethanol could vary from 58 to70 billion litres. It was thus concluded that because the assumption that 50% of all produced molasses could be

[22] An exemption was made for India, where it was assumed that 54% of all produced molasses could be used to produce bioethanol because that would result in India meeting the total domestic demand for bioethanol from "sugarcane + molasses" from molasses. Based on FAPRI (2010), India will be exporting a very small surplus of sugar in 2019/2020. The assumption was made that India would be unlikely to allow its domestic sugar production to diminish, at the risk of not being able to satisfy its domestic demand, in order to produce a small amount of bioethanol directly from sugarcane juice in order to meet its biofuel targets. Instead it was assumed that the bioethanol demand would be met through higher use of molasses ethanol.

used to produce fuel bioethanol in 2020 does not influence the order of magnitude of the demand for sugarcane juice bioethanol, the ILUC impacts are unlikely to be significantly different either.

Table 41. Total demand for bioethanol from molasses and sugarcane juice.
Note: demand shown here is for the "low demand" scenario – see following section.

Region *Country of sub region*	**Total demand for bioethanol from sugarcane + molasses in 2020 (A)**	**Demand for bioethanol from molasses in 2020 (B)**	**Demand for bioethanol from sugarcane in 2020 (C=A-B)**
	Billion litres	**Billion litres**	**Billion litres**
Africa	**0.92**	**0.79**	**0.12**
Southern Africa	*n/a*	*0.79*	*n/a*
China	**1.31**	**0.72**	**0.59**
Eastern Europe	**-**	**-**	**-**
EU	**1.54**	**-**	**1.54**
India	**2.23**	**2.23**	**-**
Latin America	**33.8**	**0.54**	**33.3**
Central America	*n/a*	*0.54*	*n/a*
Middle East	**-**	**-**	**-**
OECD North America	**21.1**	**-**	**21.1**
OECD Pacific	**5.24**	**0.17**	**5.1**
Australia	*n/a*	*0.17*	*n/a*
Other Asia	**3.22**	**1.10**	**2.1**
Indonesia	*n/a*	*0.14*	*n/a*
Pakistan	*n/a*	*0.28*	*n/a*
Philippines	*n/a*	*0.14*	*n/a*
Thailand	*n/a*	*0.44*	*n/a*
Viet Nam	*n/a*	*0.10*	*n/a*
TOTAL	**69.4**	**5.6**	**63.8**

In this chapter, bioethanol from sugarcane only refers to bioethanol made directly from sugarcane juice and does not include bioethanol that is produced from molasses.

8.2.2 Variation on the demand for sugarcane ethanol in Europe and the US

In one set of scenarios, the demand for sugarcane bioethanol in Europe was based on an assumption that only 10% of the European bioethanol would be from sugarcane (based on European Commission, 2007). In the US, it was assumed that 30% of all bioethanol would be from sugarcane. As, these demand projections were considered to be at the lower end of the range by several stakeholders, we defined an alternative biofuel projection, where 90% of the European bioethanol demand would be satisfied by sugarcane bioethanol and where the totality of the advanced biofuel target in the US would be met by sugarcane bioethanol.

Table 42 shows the breakdown of the global demand for bioethanol from sugarcane per world region in the low and the high demand scenario.

Table 42. Above baseline demand for sugarcane bioethanol in 2020 in different world regions in the low and high demand scenarios.

Region	Demand for sugarcane bioethanol in the low demand scenario	Demand for sugarcane bioethanol in the high demand scenario
	[billion litres]	[billion litres]
OECD NA	21.1	57.0
EU	1.5	13.8
Eastern Europe	-	-
China	0.59	0.59
India	-	-
Other Asia	2.11	2.11
OECD Pacific	5.07	5.07
Africa	0.12	0.12
Latin America[23]	4.97	4.97
Middle East	-	-
Total	35.5	83.7

8.3 Market responses

The additional demand for sugarcane for bioethanol could be met in a number of different ways:

1. Increased availability of sugarcane, either through substitution of cane sugar in the food market by beet sugar or other sweeteners or by increased production of sugarcane;
2. Increased availability through improvements in efficiency in the supply chain;
3. Reduction in amount of sugarcane demand in other markets.

The other market response to be considered is the effect that co-products from sugarcane bioethanol production have on the markets in which they act as substituting products.

These market responses to the increased demand for sugarcane for bioethanol are mapped out in Figure 31. The magnitude of the effect of the different market responses is discussed further in this chapter and illustrated through the scenarios.

As discussed earlier in this report, market responses 2 and 3 have not been considered in this analysis. The extent to which these market effects make a contribution could be explored in future analyses of this type.

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[23] Demand for sugarcane bioethanol in 2008 in Brazil was already high. In this analysis we only look at the impact of an additional (above baseline i.e. above 2008) demand for sugarcane. Thus the total demand for sugarcane bioethanol in Brazil in 2020 is assumed to be 33 billion litres, but the additional demand is 5 billion litres.

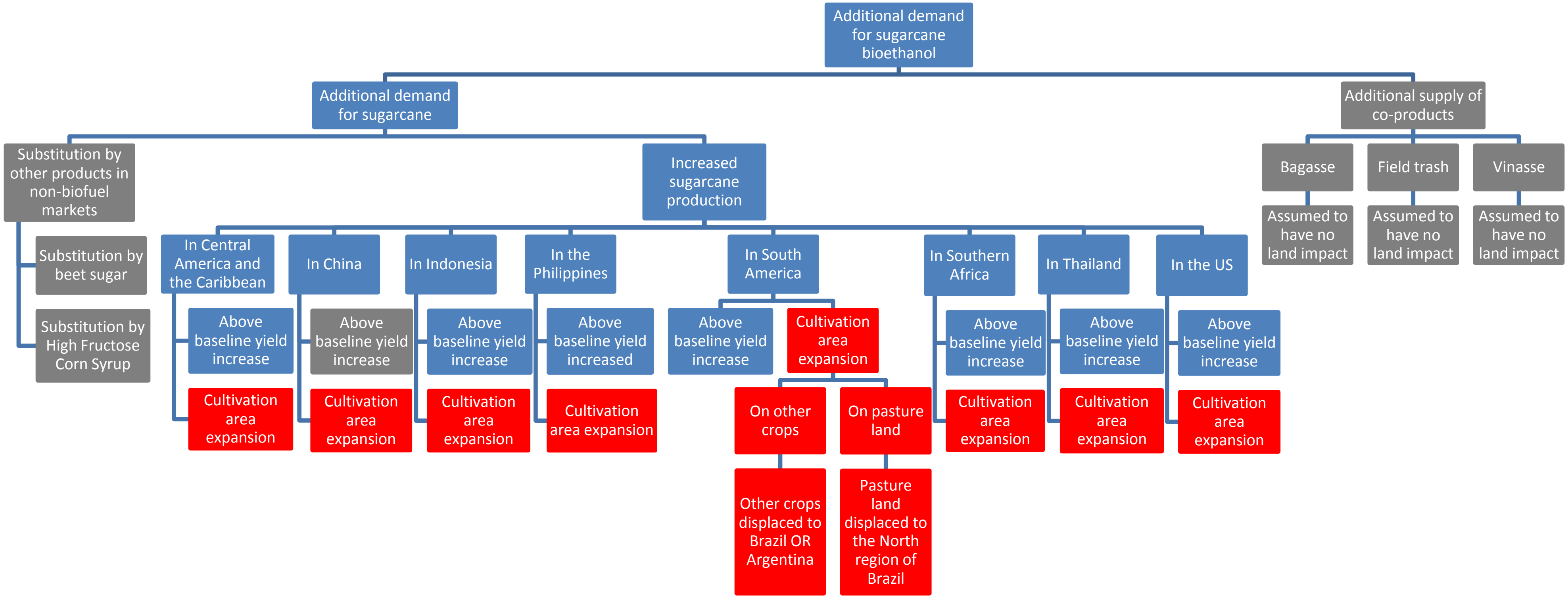

Figure 31. Market responses to an increase in demand for sugarcane bioethanol.

Land expansion, or ILUC "debits" are shaded red and avoided land expansion, or ILUC "credits" are shaded green. Market responses depicted in the diagram in grey are not considered further for reasons explained in the text.

8.3.1 *Displacement of sugarcane out of other markets*

Sugar is traded on a global commodity market and so we will consider at a global level the substitution of cane sugar in the food market by other sugars or sweeteners. However, because this market is distorted by domestic market protection policies, we take a closer look at national situations where needed.

8.3.1.1 No displacement by beet sugar

An obvious replacement for cane sugar would be beet sugar. However sugarcane is highly productive in tropical climates, where most of today's population also lives, and the production of cane sugar has thus been outgrowing the production of beet sugar over the past 3 or 4 decades (see Figure 32). This trend was assumed to continue into the future, especially with government subsidies and internal market protection for beet sugar in Europe and the US slowly disappearing.

Thus it was assumed that no cane sugar would be displaced out of the food market by beet sugar.

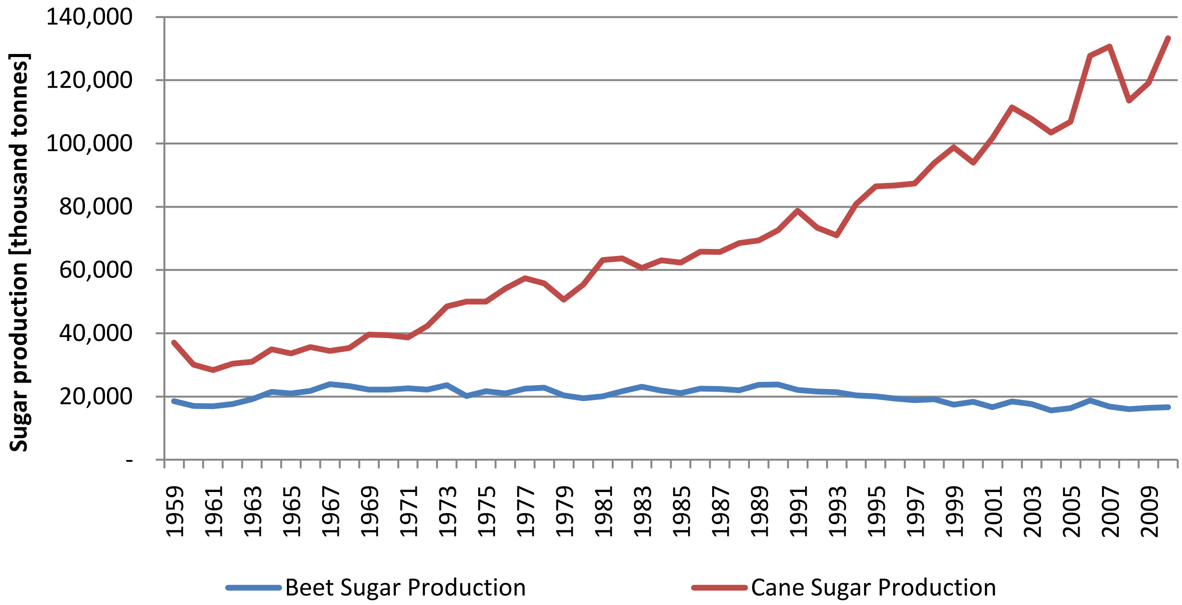

Figure 32. Historical production of sugar from sugarcane and sugar beet.

Source: USDA FAS (2010).

8.3.1.2 No displacement by other sweeteners such as High Fructose Corn Syrup

Cane sugar can also be replaced by other sweeteners, such as High Fructose Corn Syrup (HFCS). HFCS is a group of liquid sweeteners produced through enzymatic conversion of the glucose in corn syrup into fructose. The two most important HFCS products contain 42% fructose (HFCS-42) and 55% fructose (HFCS-55). HFCS was first introduced into the food and beverage industry in the US in the 1970's and its use (especially in the US) has been growing since (White, 2008).

FAPRI (2010) takes HFCS demand into account when projecting consumption of sugar to 2020 and so our baseline projection does include HFCS consumption in the US and Mexico (but assumes no widespread replacement of sugar by HFCS in Europe).

We have assumed no above baseline increase in HFCS in the biofuel scenario, especially as HFCS is produced from corn, and thus is in competition with bioethanol for corn supplies.

8.3.2 Increased production of sugarcane

The additional demand for sugarcane in the biofuel projection, to replace the sugarcane potentially diverted from the food sector to the biofuel sector in the biofuel scenarios, has to be satisfied through additional sugarcane production, as we assume there is no increased substitution of cane sugar in the food sector (as discussed in the previous section). The next sub-section looks into where this additional cane production could take place. Sub-sections 8.3.2.1 to 8.3.2.9 then take a deeper look at each of the producing regions/countries.

8.3.2.1 Where will the increased production of sugarcane take place?

The geographical location of the sugarcane production depends on several parameters that will change over time and are difficult to predict:

- the suitability of the country / region to produce sugarcane in important quantities – this depends on climate and soil conditions, as well as experience in growing the crop (Fischer et al., 2002);
- the cost of producing sugarcane and bioethanol from sugarcane in the country / region – this can vary significantly in different regions. For example, in mid-2005, sugar production costs in the three lowest cost countries were estimated at $ 145 / tonne in Brazil, $ 185 / tonne in Australia and $ 195 / tonne in Thailand. About one-quarter of the total worldwide sugar production is at $ 200-250 / tonne, above which the cost jumps to $ 400 / tonne and higher (Kojima and Johnson, 2006);
- the potential for scaling up production – land, infrastructure and technology availability are other important factors for countries / regions to increase production.

In this study, we did not carry out a detailed analysis of each country's potential to increase sugarcane cultivation and to produce bioethanol. Consideration of the factors above and stakeholders' views on their likely future evolution in different world regions have informed our analysis. The main conclusions of our analysis area:

- Brazil will remain the main producer of sugarcane bioethanol in 2020, through the growth in demand in its domestic market and increased exports. South America as a whole will increase its production of bioethanol from sugarcane. Apart from satisfying its internal demand, South America will export to Europe, OECD North America, OECD Pacific and Other Asia.
- Central America and the Caribbean are likely to increase their current production of bioethanol, to supply both OECD North America and OECD Pacific. Due to advantageous import tariffs, some sugarcane bioethanol produced in Brazil can be export to the US via the Caribbean islands. However, in our analysis, this would be considered to come from Brazil as we are concerned with the location and therefore carbon stock of the land on which the cane will actually be grown.
- China will supply its own needs in terms of sugarcane bioethanol. Given its prior stance of not permitting the use of food crops for biofuels (Qiu et al., 2010), China is considered more

likely to scrap future bioethanol targets than divert food crops to biofuel or import biofuels in order to meet biofuel targets.

- Several countries in Asia are considered to have a high suitability for sugarcane production, although land availability can become a problem in these countries. The countries considered in this study were Indonesia, the Philippines and Thailand (Wik et al., 2008; Thailand's Public Relations Department, 2008). These countries were considered to supply bioethanol to their domestic markets and also to export to OECD Pacific.
- Southern Africa is often cited as a region with a significant sugarcane production potential (Wik et al., 2008). However, infrastructure and institutional barriers to scaling up in these countries need to be factored in, as well as the potentially significant need for irrigation. To represent the uncertainty around the Southern African production potential, we consider two different scenarios: (a) Southern Africa only produces bioethanol for consumption in South Africa; (b) Southern Africa produces bioethanol both for domestic consumption and for export to Europe.
- Finally, the United States has some potential for sugarcane production, especially in the southern states. However, sugarcane growing has been largely abandoned in the US. Some stakeholders expressed the view that farmers in the United States could restart sugarcane production if they saw renewed interest in the crop for bioethanol production. Others were very sceptical. Two different scenarios are defined around the contribution of the US to meeting its domestic demand for sugarcane bioethanol.
- The OECD Pacific region includes countries such as Australia, New Zealand, Japan, etc. which have targets for sugarcane bioethanol. However, most of these countries do not have the potential for producing much additional sugarcane. For example, the production of Japan is highly constrained by land availability – in fact Japan is already a sugar importer. Australia on the other hand is producing important amounts of sugar, but considering water constraints, it is unlikely to be able to scale up production significantly. It was thus considered that this region would import all the sugarcane bioethanol it needs – from regions such as South America (Japan has already invested in sugarcane mills in Brazil) and Other Asia (due to the geographical proximity).

Combining the two different biofuel demand projections (cf. section 8.2.2) and our assumptions around sugarcane supply, three sets of scenarios were defined:

1. low demand scenario + some production in the United States + Southern African production for export (called "low demand with US and SA supply");
2. low demand scenario + no production in the US + no Southern African export (called "low demand with no US and SA supply");
3. high demand scenario + some production in the United States + Southern Africa production for export (called "high demand with US and SA supply").

Table 43 to Table 45 present demand and supply locations (in the rows and columns, respectively) for each of the three scenarios described above.

Table 43. Scenario with Low Demand with US and SA Supply – sugarcane bioethanol demand and supply volumes (bn litres).

Exporter / Importer	Central America	China	Indonesia	Philippines	South America	Southern Africa	Thailand	USA	Total
Africa						0.1			**0.1**
China		0.6							**0.6**
Eastern Europe									**-**
EU					1.0	0.5			**1.5**
India									**-**
Latin America					5.0				**5.0**
Middle East									**-**
Other Asia			0.3	0.3	0.9		0.6		**2.1**
OECD North America	6.3				13.7			1.1	**21.1**
OECD Pacific	1.0		0.5	0.5	1.5		1.5		**5.1**
Total	**7.3**	**0.6**	**0.8**	**0.8**	**22.1**	**0.6**	**2.2**	**1.1**	**35.5**

Table 44. Scenario with Low Demand with No US and SA Supply – sugarcane bioethanol demand and supply volumes (bn litres).

Exporter / Importer	Central America	China	Indonesia	Philippines	South America	Southern Africa	Thailand	USA	Total
Africa						0.1			**0.1**
China		0.6							**0.6**
Eastern Europe									**-**
EU					1.5				**1.5**
India									**-**
Latin America					5.0				**5.0**
Middle East									**-**
Other Asia			0.3	0.3	0.9		0.6		**2.1**
OECD North America	6.3				14.8				**21.1**
OECD Pacific	1.0		0.5	0.5	1.5		1.5		**5.1**
Total	**7.3**	**0.6**	**0.8**	**0.8**	**23.6**	**0.1**	**2.2**	**-**	**35.5**

Table 45. Scenario with High Demand With US and SA Supply – sugarcane bioethanol demand and supply volumes (bn litres).

Exporter / Importer	Central America	China	Indone-sia	Philippi-nes	South America	Southern Africa	Thailand	USA	Total
Africa						0.1			**0.1**
China		0.6							**0.6**
Eastern Europe									**-**
EU					13.3	0.5			**13.8**
India									**-**
Latin America					5.0				**5.0**
Middle East									**-**
Other Asia			0.3	0.3	0.9		0.6		**2.1**
OECD North America	11.4				43.3			2.1	**56.8**
OECD Pacific	1.0		0.5	0.5	1.5		1.5		**5.1**
Total	**12.4**	**0.6**	**0.8**	**0.8**	**64.0**	**0.6**	**2.2**	**2.1**	**83.5**

South America is by far the largest producer and exporter of sugarcane bioethanol in 2020. Central America and the Caribbean increases its role as an exporter of bioethanol, mainly to North America. Other regions or countries have much smaller productions, usually for both domestic consumption and exports.

The production projections assumed for South America in this study are consistent with other projections. For example, the União da Indústria de Cana-de-açúcar (UNICA) (2008a) projects that Brazil alone will have 15.7 billion litres of bioethanol available for export by 2020 – where we assumed South America would export between 17 and 19 billion litres in the low demand scenarios.

A final parameter that determines the additional demand for sugarcane is the conversion efficiency of sugarcane to bioethanol. The conversion efficiency depends on two independent parameters:

- cane quality, or sucrose content;
- and efficiency in sucrose extraction.

Cane quality depends on the time at which the sugarcane is harvested, the climatic conditions right before the harvest and on the variety. Brazil has been investing heavily in research into new enhanced varieties for higher cane quality. A steady increase in cane quality is expected over the next 15 years (Macedo et al., 2008). However, the sucrose extraction efficiency is currently already very high, around 90%, and thus higher efficiencies will be difficult to achieve. Overall, Macedo et al. (2008) estimate that conversion efficiency in Brazil would increase from 86.3 L ethanol / tonne sugarcane in 2005/2006 to 92.3 L ethanol / tonne sugarcane in 2020.

In other producing regions, the industrial efficiency is likely to achieve the same level of sucrose recovery in 2020 than Brazil. However, the quality achieved will likely be lower than in Brazil – this would lead to a lower conversion efficiency overall.

We carried out a sensitivity test where the conversion efficiency in OECD North America and Latin America was set to be equal to that in Brazil in 2020 and the conversion efficiency in the rest of the world was 10% lower (i.e. 83.1 L ethanol / tonne sugarcane). This led to a change of approximately 1% in the final ILUC factor. Therefore, the conversion efficiency was kept constant over all world regions (i.e. at the Brazilian conversion efficiency).

The following sections explore in more detail the yield increase and cultivation area expansion associated with the projected additional production of sugarcane.

8.3.2.2 Increased production in Central America and the Caribbean

Central America and the Caribbean[24] have quite a high suitability for sugarcane cultivation combined with land availability. This region could gain from the Brazilian experience, especially in terms of varieties, harvest practices, and conversion technologies. Some countries, such as Guatemala, are already emerging as the leading sugarcane producers in the region.

The total additional production of sugarcane needed from this region varies between 80 and 134 million tonnes, depending on the scenario considered. This is to be compared with a total production of 106 million tonnes for food purposes projected by FAPRI (2010) in 2020. To determine how this additional production would be met – through above baseline yield increase and/or cultivation area expansion – we applied the methodology developed by Lywood et al. (2009a) and described in section 3.3.2.2.

Lywood et al. (2009a) did not provide a statistical analysis of the relationship between yield and area growth contribution to additional production for sugarcane in Central America and the Caribbean, but by replicating the methodology, such a relationship was estimated. The Lywood et al. (2009a) methodology provided a good fit between model output and historical data – better than considering a constant yield improvement. For example, when comparing results for the previous 15-year period (from 1991 to 2007), the Lywood et al. (2009a) methodology predicts a yield CAGR of 0.7% whereas a constant yield improvement method would have resulted in a yield CAGR of 1.1% – the actual yield CAGR being 0.2%.

Table 46 presents the projected sugarcane yield and cultivation area for the different supply scenarios for Central America and the Caribbean.

[24] Central America includes all countries that lie South of Mexico and North of Colombia, i.e. Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua and Panama.

Caribbean refers to all the Caribbean Islands, i.e. Anguilla, Antigua and Barbuda, Aruba, Bahamas, Barbados, British Virgin Islands, Cayman Islands, Cuba, Dominica, Dominican Republic, Grenada, Guadeloupe, Haiti, Jamaica, Martinique, Montserrat, Netherlands Antilles, Puerto Rico, Saint Barthélemy, Saint Kitts and Nevis, Saint Lucia, Saint Martin, Saint Vincent and the Grenadines, Trinidad and Tobago, Turks and Caicos Islands, Unites States Virgin Islands.

Table 46. Overview of production area and yield projections for sugarcane in Central America and the Caribbean in the baseline and biofuel projections in 2008 and 2020.

Year	2008	2020			
Projection		Baseline	Biofuels		
ILUC scenario			1-3, 5, 7-9, 12 and 13	4, 6, 10 and 11	14
Additional production due to biofuels [million tonnes]	n/a	n/a	79.5	134	79.5
Total production [million tonnes]	82.9	106	186	241	186
Total harvested area [million ha]	1.3	1.5	2.5	3.0	2.7
Yield [tonne / ha]	63.6	69.6	75.7	79.8	69.6

In most of the scenarios, we expect an additional ~0.9 million hectares to come into production by 2020 due to demand for biofuels. This corresponds to a compound annual growth rate of 5.4%, which can be seen as quite optimistic to achieve. On the other hand, Fischer et al. (2002) estimated that the Central America and Caribbean region has 26 million hectares of land suitable for sugarcane cultivation.

The high demand scenarios (i.e. ILUC scenarios 4, 6, 10 and 11) project an even higher increase in cultivation area. However, this is an extreme scenario where both the EU and the US would get most of their bioethanol from sugarcane. If this is to be achieved, we would expect high levels of investment to be directed into production facilities in countries such as Central America with high production potential.

Scenario 14 projects an increase of 2.7 million hectares. Scenario 14 was carried out to assess the influence of the assumed above-baseline yield increases on the final ILUC factor. Therefore, it represents a worst case scenario as all additional sugarcane production would come from area expansion.

8.3.2.3 Increased production in China

China is expected to supply its own domestic demand for bioethanol through a combination of bioethanol from molasses, sugarcane and cassava. In this chapter, we look at the additional demand for sugarcane bioethanol. In all ILUC scenarios, the Chinese additional demand for sugarcane for the biofuel market is 6 million tonnes, compared to a production of 141 million tonnes for the food market in 2020 (FAPRI, 2010). We therefore decided to keep the yield constant and equal to the 2020 yield projected by FAPRI (2010), as the additional demand for sugarcane for biofuel was considered not to be high enough to result in above baseline yield increases. All additional sugarcane production would come from cultivation area expansion.

Table 47 below presents the production, yield and cultivation area projections for China.

Table 47. Overview of production area and yield projections for sugarcane in China in the baseline and biofuel projections in 2008 and 2020.

Year	2008	2020	
Projection	All	Baseline	Biofuels
ILUC scenario	All	All	All
Additional production due to biofuels [million tonnes]	n/a	n/a	6.42
Total production [million tonnes]	124	141	147
Total harvested area [million ha]	1.7	1.8	1.9
Yield [tonne / ha]	71.2	76.9	76.9

Fischer et al. (2002) reports that China has about 4 million hectares of rain-fed land suited for sugarcane cultivation. However, land availability is an issue in China, as crop cultivation is already using up most of the suitable land (Fischer et al., 2002). This is why Chinese sugarcane bioethanol production was only considered to be used for domestic consumption and why it would not exceed the small increase shown in Table 47.

8.3.2.4 Increased production in Indonesia

Indonesia is one of the countries in Asia with a very high potential for sugarcane cultivation. For example, Fischer et al. (2002) estimate that Indonesia has almost 30 million hectares of suitable rain-fed land for sugarcane cultivation. It is therefore considered in this study as a supplier of bioethanol from sugarcane to both the Other Asia region and the OECD Pacific.

The Lywood et al. (2009a) methodology was used to determine the relationship between yield and area growth contribution to increased production but did not provide a good fit for Indonesia. This model predicted an average yield CAGR from 1991 to 2007 of 7.3% while historical data shows a CAGR of -0.4%. Thus this methodology was not used to project yield and area contribution to the additional production of sugarcane.

Based on FAPRI (2010) projections of yield, area and production of sugarcane between 2008 and 2020, we estimated that FAPRI projects that 94% of the additional production of sugarcane for food in 2020 would come from yield increases while only 6% from area expansion[25]. If these percentages were used in the biofuel projection, sugarcane yields in Indonesia would increase from 70 t / ha in 2008 to 98 t / ha in 2020 (in the case of scenario 1) – i.e. the 2020 yield would be higher than in Brazil. These results were considered unrealistic and so the FAPRI-based percentages were not used for the Indonesian biofuels projection.

[25] This high yield contribution can be explained by the fact that FAPRI (2010) projects a high yield increase in Indonesia (from 67 t / ha to 71 t / ha) between 2008 and 2020 and a low increase in production (from 26 to 28 million tonnes sugarcane).

As can be seen on Figure 33, the historical evolution of sugarcane yields in Indonesia is surprising. From a very high level in the beginning of the 70's it has been decreasing to very low yields in 2008[26]. However, this decreasing trend has been tailing off over the past 2 decades. We have assumed in our analysis that the historical trend in sugarcane yields in Indonesia will continue in the baseline – i.e. yields in Indonesia will stay constant at current levels out to 2020 in the baseline projection. In the biofuel projection, we assume a small increase in yields, based on the fact that the additional demand is substantial (~32% of demand for food) – i.e. yields will increase by 1%/year in the biofuel projection.

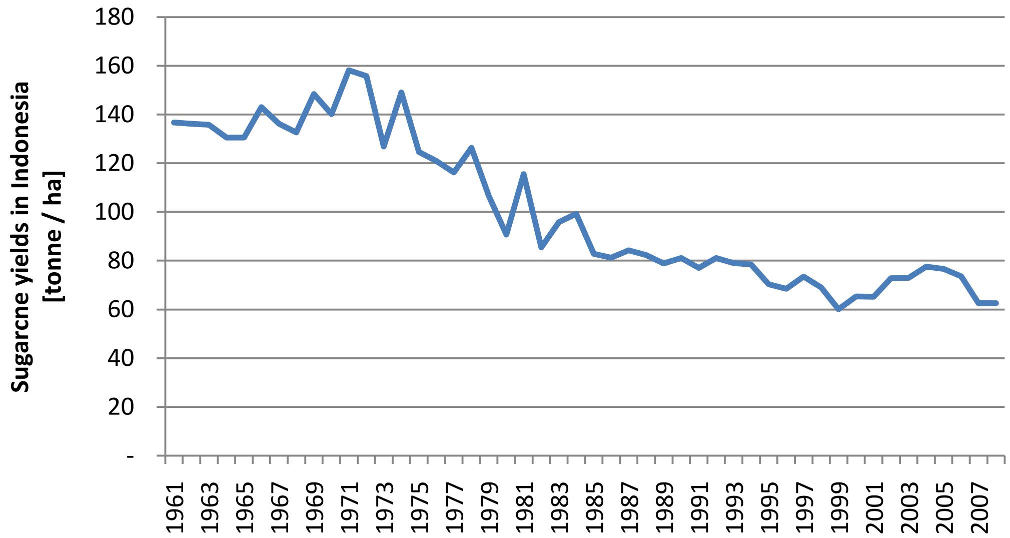

Figure 33. Historical evolution of sugarcane yields in Indonesia since 1961.

Source: FAO, 2010a.

FAO (2010a) shows sugarcane yield for Indonesia at 62 t / ha in 2008. FAPRI (2010) presents a yield of 73 t / ha. A yield of 70 t / ha was used in this study as representative of the two. Projections of production, harvested area and yield for the baseline and biofuel projection are presented in Table 48 below.

[26] Several reasons for decreasing yields can be put forward that are more or less important in different world regions, such as climatic conditions (and especially occurrence of storms, typhoons, etc.), weather patterns (e.g. droughts, La Niña and El Niño effects, etc.), political and institution instability, poor law enforcement around private property and thus land changing hands often, etc.

Table 48. Overview of production area and yield projections for sugarcane in Indonesia in the baseline and biofuel projections in 2008 and 2020.

Year	2008	2020		
Projection		Baseline	Biofuels	
ILUC scenario			1-13	14
Additional production due to biofuels [million tonnes]	n/a	n/a	8.93	8.93
Total production [million tonnes]	25.6	27.6	36.5	36.5
Total harvested area [million ha]	0.36	0.39	0.46	0.52
Yield [tonne / ha]	70.0	70.0	78.9	70.0

8.3.2.5 Increased production in the Philippines

The Philippines is another Asian country considered to have a high potential for sugarcane production. Currently, the Philippines have a bioethanol target, and part of that bioethanol is expected to be produced from sugarcane (Lotilla, 2007).

The Lywood et al. (2009a) methodology or FAPRI (2010) projections were not used to project yield and area growth contribution to additional production of sugarcane for the same reasons described for Indonesia (as described above). The Lywood et al. (2009a) methodology predicts an average yield CAGR of 0.3% between 1991 and 2007 while historical data shows a CAGR of 1.0%. The same approach as for Indonesia was thus followed, i.e. assumptions on continued historical trends in sugarcane yield in the baseline projection and small increase in yields in the biofuel projection were used.

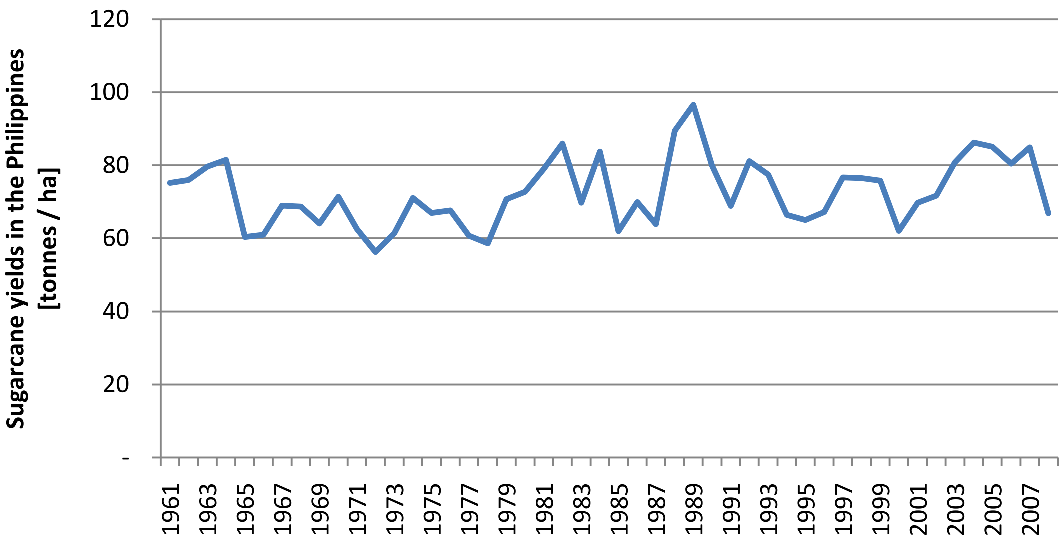

Figure 34. Historical evolution of sugarcane yields in the Philippines since 1961.

Source: FAO, 2010a.

As can be seen in Figure 34, sugarcane yields have been varying around 80 t / ha for several decades. It was therefore assumed that, in the baseline projection, yields would stay constant at current levels out to 2020. FAO (2010a) estimate of yield in 2008 was 67 t / ha and FAPRI (2010) used a yield of 56 t / ha. In this study, a yield of 60 t / ha was assumed. In the biofuels projection, however, an increase of 1%/year was assumed. Table 49 summarises the production, area and yield projection for the Philippines.

Table 49. Overview of production area and yield projections for sugarcane in the Philippines in the baseline and biofuel projections in 2008 and 2020.

Year	2008	2020		
Projection		**Baseline**	**Biofuels**	
ILUC scenario			**1-13**	**14**
Additional production due to biofuels [million tonnes]	n/a	n/a	8.93	8.93
Total production [million tonnes]	21.6	27.6	36.5	36.5
Total harvested area [million ha]	0.36	0.46	0.54	0.61
Yield [tonne / ha]	60.0	60.0	67.6	60.0

8.3.2.6 Increased production in South America

South America[27], and especially Brazil, is and will remain by far the biggest producer of bioethanol from sugarcane to 2020. South American production will cover its own domestic needs plus export bioethanol to OECD North America, the EU, Other Asia and OECD Pacific.

Given the prominent situation of Brazil as the main ethanol producer in South America (about 88% of all sugarcane produced in South America in 2008 was produced in Brazil), Brazil was taken as a proxy for South America for yield and area projections. We thus used the contribution of yield increase and area expansion established for Brazil by Lywood et al. (2009a) when projecting yield increase and area expansion for South America. The results are present in the table below.

[27] South America includes all countries located South of Panama, i.e. Argentina, Bolivia, Brazil, Chile, Colombia, Ecuador, Falkland Islands, French Guiana, Guyana, Paraguay, Peru, South Georgia and South Sandwich Islands, Suriname, Uruguay and Venezuela.

Table 50. Overview of production area and yield projections for sugarcane in South America in the baseline and biofuel projections in 2008 and 2020.

Year	2008	2020				
Projection		**Baseline**	**Biofuels**			
ILUC scenario			**1, 5, 7-9, 12 & 13**	**2 & 3**	**4, 6 & 10-11**	**14**
Additional production due to biofuels [million tonnes]	n/a	n/a	239	256	693	239
Total production [million tonnes]	758	784	1.02	1.04	1.48	1.02
Total harvested area [million ha]	9.7	9.6	12	12	15	13
Yield [tonne / ha]	78.4	81.3	87.3	87.7	98.4	81.3

Yield improvements. Figure 35 shows the historical evolution of sugarcane yield in Brazil. The average CAGR is high at 1.3% over the period 1961 to 2008. As a comparison, the yield projections presented in Table 50 above correspond to CAGR of between 0.3% and 1.9% from 2008 to 2020.

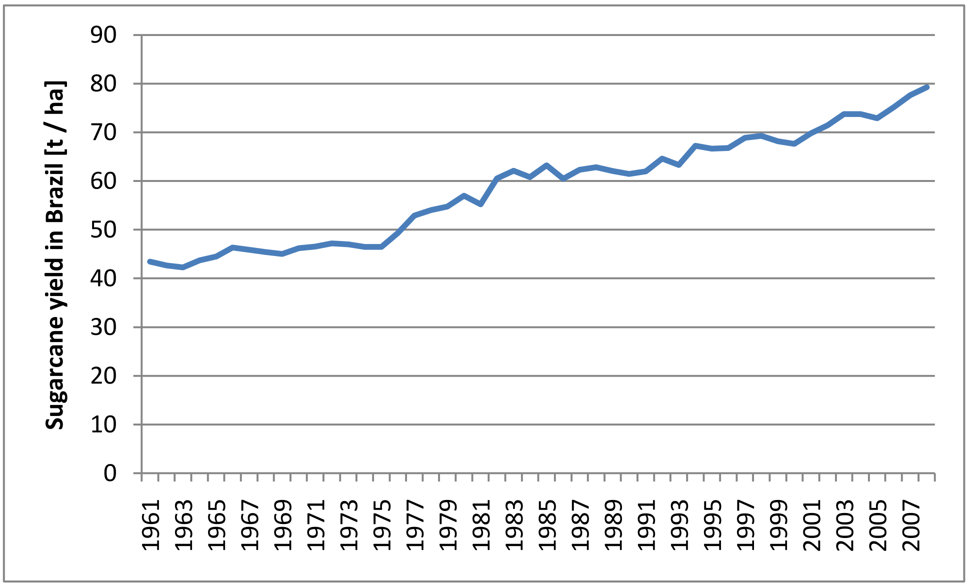

Figure 35. Historical evolution of sugarcane yield in Brazil since 1961.

Source: FAOSTAT (2010a).

Other studies have projected a continued increased in sugarcane yields in Brazil to 2020. Macedo et al. (2008) project yields of 95.0 tonne sugarcane / ha in 2020. Goldemberg (2008) foresees an increase of 12% in sugarcane yield over the next ten years in the State of São Paulo, which is where most of the sugarcane is produced in Brazil. Thus the projected yields seem reasonable in light of other Brazilian studies.

Land availability. According to Fischer et al. (2002) South America has a significant amount of land suitable for sugarcane production (318 million hectares). Although the actual amount of land available for growing sugarcane is likely to be smaller than this[28] there is still likely to be more than enough available land to meet the additional areas required in the biofuel projection. UNICA (2008a) projects that sugarcane planted area will reach 14 million hectares by 2020 – i.e. a harvested area of approximately 12 million hectares[29]. This is in line with most of our ILUC scenarios. In the high demand scenario, more pressure will be put on South America to produce more bioethanol and so it is likely that more land will be planted over a short period of time. Therefore the cultivation area expansion projected in Table 50 seems reasonable.

8.3.2.7 Increased production in Southern Africa

The potential of Southern Africa[30] to become an exporter of sugarcane bioethanol was widely discussed during the project. On the one hand, Southern Africa has the right climate and could achieve very high sugarcane yields (see Figure 36). Furthermore, many Southern African countries have a significant potential for sugarcane expansion – this includes countries such as Angola, Mozambique, Madagascar, Tanzania, Zambia (E4tech, 2006). But on the other hand, big barriers exist to the achievement of this potential: political and institutional instability, infrastructure problems, especially for land locked countries, etc.

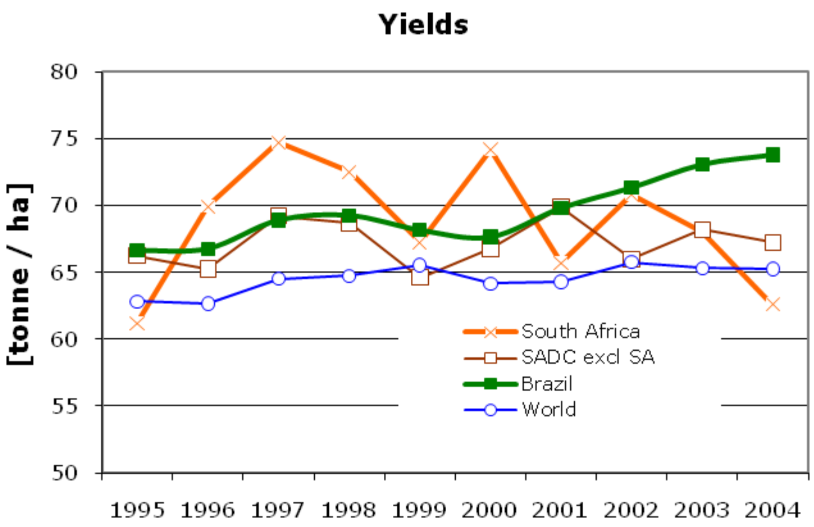

Figure 36. Sugarcane yields in South Africa and in SADC excluding South Africa, compared to historical yields in Brazil and in the world.

Source: Based on FAO statistics from 2005.

[28]Fischer et al. (2002) estimated the suitable land for rain-fed sugarcane production. For Brazil, this amounted to 318 million hectares. However, Fischer et al. (2002) did not account for suitable land that already has an existing productive use, such as natural habitat, other agriculture, human settlements and infrastructure etc. But based on the fact that the amount of suitable land is an order of magnitude bigger than the needed planted area of sugarcane, we assume enough currently “unused” land would be available.

[29] See section 8.3.2.10 for a description of the difference between planted and harvested area.

[30] In this study, we have taken Southern Africa to be the countries that are members of the Southern African Development Community (SADC), i.e. including Angola, Botswana, Democratic Republic of the Congo, Lesotho, Madagascar, Malawi, Mauritius, Mozambique, Namibia, South Africa, Swaziland, Tanzania, Zambia and Zimbabwe.

As can be seen from Figure 36, it is difficult to determine an average yield of sugarcane in 2020 based on historical trends as no historical trend appears clearly. According to Schut et al. (2010), the average expected yields by the 3 biggest sugarcane projects that were proposed to the Mozambique Government (before 2008) were 113.3 tonne sugarcane / ha. However Schut et al. (2010) also say that by comparison, the best average yield for the Mozambican industry over the past 5 years was 72 tonne sugarcane / ha and the best average company yield over the same period was 87 tonne sugarcane / ha.

Other studies suggest typical current yields around 65 to 67 tonne sugarcane / ha have stayed constant for the past 10 to 15 years (Bezuidenhout, 2007; Tongaat Hullet Sugar, 2010; TSB Sugar, 2010; South African Sugar Association, 2010) while studies incorporating demand for bioethanol from sugarcane tend to look at yield increases up to 90+ tonnes sugarcane / ha by 2020/2025 (Johnson and Matsika, 2006).

Therefore, the average yield for Southern Africa was taken to be a conservative 67 tonnes / ha in 2008 and to stay constant to 2020 in the baseline projection. Then, in the biofuels projection, yields were assumed to grow by 1% each year to 2020, which is a fairly conservative assumption. Table 51 summarises the yield increase and area expansion projections for Southern Africa.

Table 51. Overview of production area and yield projections for sugarcane in Southern Africa in the baseline and biofuel projections in 2008 and 2020.

Year	**2008**	**2020**				
Projection		**Baseline**	**Biofuels**			
ILUC scenario			**1, 5, 7-9, 12 & 13**	**2 & 3**	**4, 6 & 10-11**	**14**
Additional production due to biofuels [million tonnes]	n/a	n/a	6.75	1.33	6.75	6.75
Total production [million tonnes]	13.9	16.6	23.4	18.0	23.4	23.4
Total harvested area [million ha]	0.21	0.25	0.31	0.27	0.31	0.35
Yield [tonne / ha]	67.0	67.0	75.5	67.0	75.5	67.0

8.3.2.8 Increased production in Thailand

Thailand is currently producing most of its bioethanol from molasses. However, with the introduction of a bioethanol target and government support for biofuel production (including the Ministry of Industry's Action Plan on Sugar Cane Development, bioethanol production from sugarcane is likely to increase (Nguyen et al., 2008; Thailand's Public Relation Department, 2008). We have assumed that Thailand would produce sugarcane bioethanol both for domestic use in the Other Asia region and for export to OECD Pacific.

As for Indonesia, the Philippines and Southern Africa, the Lywood et al. (2009a) methodology did not show a very good fit between its results and historical trends in yields and area expansion. Again the model predicted an average yield CAGR from 1991 to 2007 of 0.5% while historical data shows a

CAGR of 1.0%. We thus used the same approach as for the three regions cited above. Figure 37 shows the historical evolution of sugarcane yield in Thailand.

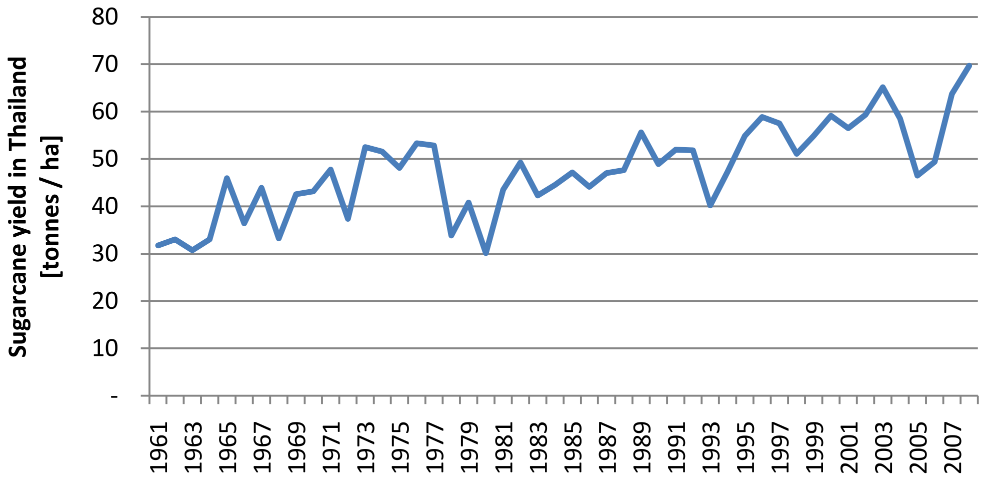

Figure 37. Historical evolution of sugarcane yields in Thailand since 1961.

Source: FAO, 2010a.

Between 1961 and 2008, yields in Thailand increased by an average of 2%/ year. If such a trend was extrapolated to 2020, the yields in Thailand would be higher than in Brazil. We therefore limited the growth to 1%/year in the baseline projection. In the biofuel projection, we assumed an increase of 1.5%/yr. FAO (2010a) reports a yield of 70 t / ha in Thailand in 2008 and FAPRI (2010) a yield of 67 t / ha. The latter number was used as the 2008 sugarcane yield in Thailand.

Table 52. Overview of production area and yield projections for sugarcane in Thailand in the baseline and biofuel projections in 2008 and 2020.

Year	2008	2020		
Projection		**Baseline**	**Biofuels**	
ILUC scenario			**1-13**	**14**
Additional production due to biofuels [million tonnes]	n/a	n/a	23.3	23.3
Total production [million tonnes]	66.5	86.5	110	110
Total harvested area [million ha]	1.0	1.2	1.4	1.5
Yield [tonne / ha]	66.5	74.9	79.5	74.9

Fischer et al. (2002) estimate that Thailand has about 3 million hectares of suitable rain-fed land for sugarcane cultivation which would suggest that the increase from 1 to 1.5 million hectares in the biofuel projection from 2008 to 2020 is feasible.

8.3.2.9 Increased production in the USA

There was much debate around whether the US would restart sugarcane cultivation to produce bioethanol domestically. Thus we considered two different cases: one in which US production would increase in the biofuel scenario and the other where it would not. We also assumed that in the case of a high demand for sugarcane bioethanol in the US, more sugarcane bioethanol would be produced domestically than in the low demand scenarios.

The same approach based on the historical trend in yield was used for the USA as for other regions such as Thailand, Indonesia, the Philippines for the same reasons. Figure 38 shows the evolution of sugarcane yield in the US. These yields have been decreasing since 1961, although at a very slow rate. In our study, it was assumed that yields would stay constant at 2008 level out to 2020 in the baseline projection but would increase by 1% in the biofuel projection when the additional production due to bioethanol demand is low and 1.5% when the additional production is high. The starting yield was taken to be FAPRI's (2010) estimate at 71.3 t / ha.

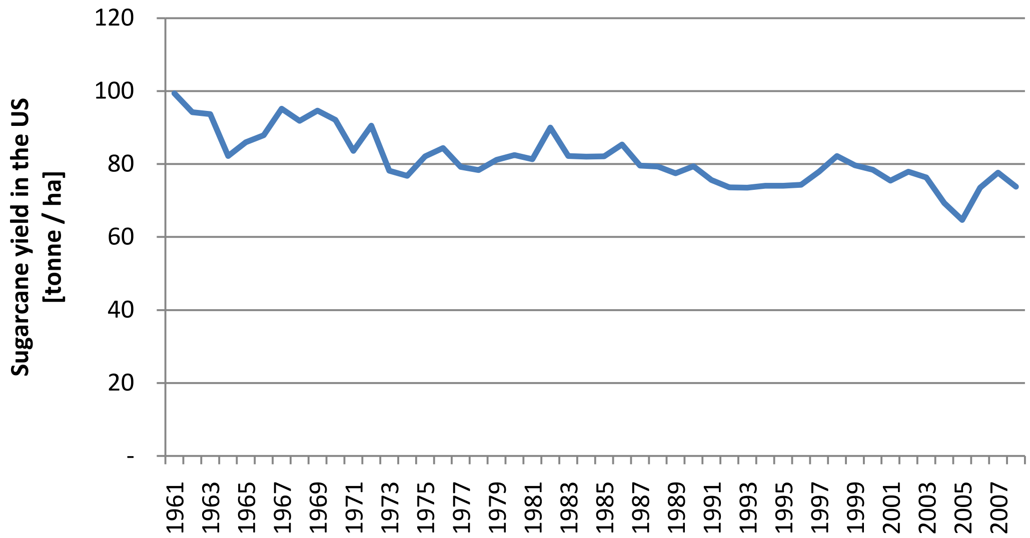

Figure 38. Historical evolution of sugarcane yields in the United States since 1961.

Source: FAO, 2010a.

Table 53 presents the results of the production, yield and area projections for the US. Overall the total harvested sugarcane area stays small (maximum 0.5 million hectares).

Table 53. Overview of production area and yield projections for sugarcane in the United States in the baseline and biofuel projections in 2008 and 2020.

Year	2008	2020				
Projection		**Baseline**	**Biofuels**			
ILUC scenario			**1, 5, 7-9, 12 & 13**	**2 & 3**	**4, 6 & 10-11**	**14**
Additional production due to biofuels [million tonnes]	n/a	n/a	11.4	-	22.8	11.4
Total production [million tonnes]	23.7	26.6	38.0	26.6	49.4	38.0
Total harvested area [million ha]	0.33	0.37	0.47	0.37	0.58	0.53
Yield [tonne / ha]	71.3	71.3	80.3	71.3	85.3	71.3

8.3.2.10 Summary

When accounting for the total area expansion due to sugarcane cultivation for bioethanol production, it is important to take into account that harvested and planted areas are different. Harvested areas represent the land that is actively producing sugarcane in a certain year. However, sugarcane is a perennial crop, which usually takes 12 to 18 months to become productive and can then be harvested over the next 5 years. Thus the actual planted area is 6/5 times the harvested area.

Table 54 summarises the results of the analysis discussed in the previous sub-sections by presenting the additional planted sugarcane area needed to fulfil the demand for sugarcane bioethanol in all producing regions under the different ILUC scenarios.

Table 54. Additional planted sugarcane area (in million hectares) in 2020 in each of the producing regions.

ILUC scenario	1, 5, 7-9, 12 & 13	2 & 3	4, 6 & 10-11	14
Central America and the Caribbean	1.11	1.11	1.79	1.37
China	0.10	0.10	0.10	0.10
Indonesia	0.08	0.08	0.08	0.15
Philippines	0.10	0.10	0.10	0.18
South America	2.51	2.67	6.45	3.53
Southern Africa	0.07	0.02	0.07	0.12
Thailand	0.27	0.27	0.27	0.37
USA	0.12	-	0.25	0.19
Total	**4.37**	**4.36**	**9.11**	**6.02**

8.3.3 Effect of co-products

The cultivation of sugarcane has two main co-products:

1. The trash which is currently burned on the field. However, in Brazil, field burning will be banned by 2020, and is already banned in some states there. The trash will then most probably be left on the field, as it has some benefit in terms of soil quality. As such it could displace some fertiliser use, but as it is not a consequence of the additional demand for sugarcane bioethanol, the reduced emissions cannot be assigned to the bioethanol. The trash is thus considered to have no indirect land use change impact.
2. The bagasse is used as fuel for process heating, potentially in combined heat and power (CHP) plants. The benefits of using bagasse as fuel for process heating would be accounted for as direct emissions, and will thus not be taken into account in the indirect impacts of sugarcane bioethanol. Bagasse has been studied as an animal feed, but it was considered that it would not be used in significant quantities for this purpose in 2020 to be accounted for in this study[31]. Thus the bagasse was assumed to have no indirect land use change impact.

Finally, the production of bioethanol from sugarcane has one main co-product: vinasse. Vinasse is a nutrient-rich product used as fertiliser. It is unlikely to have other uses, and thus it considered to have no indirect land use change impact.

Therefore, none of the sugarcane bioethanol co-products are considered to have an indirect land use change impact and there are no co-product “ILUC credits” for sugarcane ethanol.

8.4 Land use impacts and greenhouse gas consequences

Land impacts were calculated based on the difference in planted area between the baseline and the biofuel projection in 2020. The analysis carried out by Winrock International for the U.S. EPA and applied in the Renewable Fuel Standard 2010 (RFS 2) on the carbon stocked in different land use types and the associated GHG emissions from conversion from one land use type to another was used. Winrock’s approach is described in further detail in section 4.4.1 (U.S. EPA, 2010).

In the following sections we have presented the type of land use change resulting from the production of bioethanol from sugarcane in each of the producing regions and the related GHG consequences.

8.4.1 Land use change from Central American and Caribbean supply

The Central America and Caribbean region is assumed to supply an additional 80-134 million tonnes of sugarcane. The area required to meet such an increase in production was determined in section 8.3.2.2 and varies between 1.1 and 1.8 million hectares of planted sugarcane area.

Type of land expanded onto. Sugarcane cultivation area in Central America and the Caribbean can either expand onto “new” land, i.e. land that is not in agricultural production in the baseline, or onto agricultural land, i.e. where other crops are grown. In the latter case, these other crops will need to

[31] This was for a number of reasons, including the costs associated with transporting bagasse to feed markets relative to using the bagasse as process fuel on site.

be produced somewhere else as long as there is continued demand for them, i.e. on some other land, potentially displacing some other agricultural crop production. This is referred to in this chapter as the knock-on effect[32]. Where the knock-on effect will end (i.e. where some “new” land will finally be converted to crop) will determine the type of land that will be converted to agricultural production – i.e. different proportions of different types of land that will be brought into production in different countries or regions as a result of agricultural expansion. A simplifying assumption was taken in this study to assume that the “knock-on effect” would “end” in Central America/the Caribbean. We used Winrock International data on the type of land that would be converted to agricultural production (US EPA, 2010) for this region. Table 55 below summarises this data.

Table 55. Types of land use change and emission factors associated with them for perennial crops in Central America and the Caribbean.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	17%	106 t CO_2e / ha	± 43 t CO_2e / ha
Grassland	12%	-4.68 t CO_2e / ha	± 4.2 t CO_2e / ha
Mixed	30%	23.0 t CO_2e / ha	± 12 t CO_2e / ha
Savannah	26%	1.18 t CO_2e / ha	± 4.3 t CO_2e / ha
Shrub land	12%	12.9 t CO_2e / ha	± 4.8 t CO_2e / ha
Wetland	1%	4.11 t CO_2e / ha	± 4.4 t CO_2e / ha
Barren	2%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.4.2 Land use change from Chinese supply

China is assumed to supply 6.4 million tonnes of additional sugarcane. The land use impact of such an increase in production was determined in section 8.3.2.3 and corresponds to an expansion of the planted sugarcane area of about 100 thousand hectares.

Type of land expanded onto. As for the Central American and Caribbean situation, sugarcane cultivation area expansion in China will also lead to a knock-on effect. Again, the knock-on effect is assumed to end in China and Winrock International (US EPA, 2010) data is used. Table 56 below summarises this data.

[32] See section 3.3.2.2 page 20 for a discussion on the assumptions behind the knock-on effect.

Table 56. Types of land use change and emission factors associated with them for perennial crops in China. Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	6%	44.7 t CO_2e / ha	± 9.1 t CO_2e / ha
Grassland	30%	-4.43 t CO_2e / ha	± 9.1 t CO_2e / ha
Mixed	23%	10.7 t CO_2e / ha	± 9.2 t CO_2e / ha
Savannah	20%	1.62 t CO_2e / ha	± 9.1 t CO_2e / ha
Shrub land	17%	13.7 t CO_2e / ha	± 9.1 t CO_2e / ha
Wetland	1%	4.65 t CO_2e / ha	± 9.1 t CO_2e / ha
Barren	3%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.4.3 Land use change from Indonesian supply

Indonesia is assumed to supply 8.9 million tonnes of additional sugarcane. The land use impact of such an increase in production was determined in section 8.3.2.4 and corresponds to an expansion of the planted sugarcane area of between 107 and 136 thousand hectares, depending on the assumption on yield increases.

Type of land expanded onto. As for the two previous cases, sugarcane cultivation area expansion in Indonesia will lead to a knock-on effect. Again, the knock-on effect is assumed to end in Indonesia and Winrock International (US EPA, 2010) data is used. Table 57 below summarises this data.

Table 57. Types of land use change and emission factors associated with them for perennial crops in Indonesia.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	39%	154 t CO_2e / ha	± 95 t CO_2e / ha
Grassland	5%	-3.54 t CO_2e / ha	± 13 t CO_2e / ha
Mixed	29%	41.3 t CO_2e / ha	± 27 t CO_2e / ha
Savannah	22%	9.60 t CO_2e / ha	± 13 t CO_2e / ha
Shrub land	3%	21.7 t CO_2e / ha	± 13 t CO_2e / ha
Wetland	2%	12.6 t CO_2e / ha	± 13 t CO_2e / ha
Barren	0%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.4.4 Land use change from Philippines supply

The Philippines is assumed to supply 8.9 million tonnes of additional sugarcane. The land use impact of such an increase in production was determined in section 8.3.2.5 and corresponds to an expansion of the planted sugarcane area of between 125 and 159 thousand hectares, depending on the assumption about yield increases.

Type of land expanded onto. As for the previous cases, sugarcane cultivation area expansion in the Philippines will lead to a knock-on effect. Again, the knock-on effect is assumed to end in the Philippines and Winrock International (US EPA, 2010) data is used. Table 58 below summarises this data.

Table 58. Types of land use change and emission factors associated with them for perennial crops in the Philippines.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	16%	102 t CO_2e / ha	± 99 t CO_2e / ha
Grassland	5%	-4.43 t CO_2e / ha	± 15 t CO_2e / ha
Mixed	54%	22.5 t CO_2e / ha	± 27 t CO_2e / ha
Savannah	19%	1.62 t CO_2e / ha	± 15 t CO_2e / ha
Shrub land	2%	13.7 t CO_2e / ha	± 15 t CO_2e / ha
Wetland	3%	4.65 t CO_2e / ha	± 15 t CO_2e / ha
Barren	0%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.4.5 Land use change from South America supply

South America is assumed to supply most of the additional sugarcane needed to fulfil the demand for bioethanol. This corresponds to between 239 and 693 million tonnes of additional sugarcane in 2020, leading to an expansion of the planted sugarcane area of between 2.5 and 6.4 million hectares (see section 8.3.2.6).

The vast majority of this production will come from Brazil. Based on FAPRI (2010) production projections, Brazil may represent about 88% of South American sugarcane production for sugar. We assumed that the same percentages as for sugar production apply to additional production of sugarcane in Brazil and the rest of South America for biofuel production.

In the next sub-section, the specific assumptions and scenarios on land use change in Brazil are described. Then in section 8.4.5.2 the situation in the rest of South America is presented.

8.4.5.1 Land use change in Brazil

Avoided reversion of sugarcane area. In the baseline projection, the area under sugarcane cultivation is slightly decreasing (by 33 thousand ha) for two reasons: (i) a low increase in the demand for sugarcane in the baseline and (ii) a high yield increase. The additional demand for sugarcane in the biofuel projection is reversing this decrease. Thus two land use change impacts should be taken into account for Brazil:

1. the avoided reversion of 33 thousand ha due to the no longer decreasing sugarcane cultivation area;
2. the actual expansion of 2.2 to 5.6 million ha (depending on the scenario considered) of sugarcane cultivation area.

Table 59 below shows the data used to assess the GHG consequences associated with the avoided reversion in Brazil. The following paragraphs then detail how the GHG consequences associated with actual sugarcane cultivation expansions were calculated.

Table 59. Type of land that would have replaced sugarcane cultivation in Brazil and the reversion factors associated with them.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Reversion data	
		30 year reversion factors	95% confidence interval
Forest	32%	87.5 t CO_2e / ha	± 78 t CO_2e / ha
Grassland	9%	-1.10 t CO_2e / ha	± 11 t CO_2e / ha
Mixed	15%	23.5 t CO_2e / ha	± 26 t CO_2e / ha
Savannah	39%	7.70 t CO_2e / ha	± 7.7 t CO_2e / ha
Shrub land	5%	25.3 t CO_2e / ha	± 6.0 t CO_2e / ha

Expansion of sugarcane cultivation area. In Brazil, the bulk of the sugarcane production is in the Centre-South region[33] of Brazil (87% in 2007), 60% of total production being in the state of São Paulo. In the Centre-South region, the growth of sugarcane production from 2006 to 2007 occurred mainly on former pasturelands (66%), while sugarcane also displaced soybeans (18%), corn (5.3%) and orange (5%). Less than 9,000 hectares not previously used were incorporated to sugarcane production (1.4% of the enlargement) (Walter et al., 2008).

In this analysis, it was thus assumed that 66% of sugarcane cultivation area in Brazil expands onto pasture land and 34% onto other, annual, crops. Table 60 presents the GHG consequences associated with these land use changes. The following paragraphs then describe the induced land use change that happens due to the displacement of either annual crops (called *knock-on effect*) or pasture (called *pasture displacement*). Finally, some scenarios around the rate of pasture intensification or deforestation in Brazil are analysed.

Table 60. Emission factors associated with the land use change from annual crop or pasture to sugarcane in Brazil.
Based on Winrock data for the US EPA (2010).

Type of land	Conversion data	
	30 year conversion factors	95% confidence interval
Annual crop	-30.8 t CO_2e / ha	± 16 t CO_2e / ha
Pasture	-1.10 t CO_2e / ha	± 4.2 t CO_2e / ha

Knock-on effect. As Brazil is the dominant country in terms of sugarcane production and expansion, the knock-on effect is treated slightly differently for Brazil than for other countries. In the previous cases, we considered the country or region of sugarcane production would also be the country

[33] The Centre-South region includes the following states: Espirito Santo, Goiás, Minas Gerais, Mato Grosso, Mato Gross do Sul, Paraná, Rio de Janeiro, Rio Grande do Sul, Santa Catarina and São Paulo.

where other crop production would be displaced to. However, as agricultural commodity markets are global, there is actually large uncertainty around the country in which the land being brought into production will be located. In the case of Brazil, we have examined two different cases:

1. All land brought into production is located in Brazil. See Table 61 for the land conversion data and emission factors used.
2. Based on Walter et al. (2008), 63% of the crops onto which sugarcane expands are soybean. We assume that all soybean cultivation area is expanding in Argentina (see Table 62 for land conversion data). The rest of the displaced cropland leads to land use change in Brazil.

Table 61. Types of land use change and emission factors associated with them for annual crops in Brazil. Based on Winrock data for the US EPA (2010).

Type of land	Share *(lower deforestation rate scenario)*	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	19% *(10%)*	131 t CO_2e / ha	± 108 t CO_2e / ha
Grassland	18% *(23%)*	30.6 t CO_2e / ha	± 18 t CO_2e / ha
Mixed	20% *(20%)*	57.6 t CO_2e / ha	± 31 t CO_2e / ha
Savannah	35% *(40%)*	39.7 t CO_2e / ha	± 16 t CO_2e / ha
Shrub land	6% *(6%)*	58.9 t CO_2e / ha	± 14 t CO_2e / ha
Wetland	0% *(0%)*	44.7 t CO_2e / ha	± 15 t CO_2e / ha
Barren	0% *(0%)*	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

Table 62. Types of land use change and emission factors associated with them for annual crops in Argentina. Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	12%	61.0 t CO_2e / ha	± 82 t CO_2e / ha
Grassland	26%	11.3 t CO_2e / ha	± 54 t CO_2e / ha
Mixed	27%	22.6 t CO_2e / ha	± 46 t CO_2e / ha
Savannah	17%	14.4 t CO_2e / ha	± 40 t CO_2e / ha
Shrub land	14%	20.6 t CO_2e / ha	± 41 t CO_2e / ha
Wetland	1%	15.9 t CO_2e / ha	± 40 t CO_2e / ha
Barren	3%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

Pasture displacement. 66% of additional sugarcane cultivation area is assumed to expand onto pasture land. While total land area used for pasture in Brazil is diminishing – especially in the Southern region of Brazil – the total cattle herd and meat production in Brazil is increasing. Therefore, pasture activity is intensifying in Brazil. Productivity can improve significantly because there are very low stocking rates in Brazil on the extensive tracts of land occupied by pastures (159 million hectares in 2006). According to the 2006 Agricultural Census, the Brazilian official average is one animal per hectare. However, in spite of this intensification potential, pasture intensification is happening very slowly and pasture continues to expand in the regions where land is cheaper, i.e. at the agricultural frontier (Schlesinger, 2010).

Therefore a very important question for our study is whether there is a causality link between sugarcane expansion and pasture displacement in the Centre-South region and pasture expansion in the North region. In other words, is it the expansion of sugarcane in the Centre-South pushing pasture land out of that region and into the North? Or are other factors causing this trend towards pasture expansion in the North, e.g. Amazonian development, regional subsidies to cattle ranching, expansion of other crops (e.g. soy), and are sugarcane farmers just being opportunistic in moving into the old pasture land in the Centre South?

No consensus has been reached on this question, as it has been difficult to find evidence that one activity is directly causing another:

- Walter et al. (2008) argue that "there is no correlation between the enlargement of sugarcane area from 1996 to 2006 mainly in São Paulo state, and deforestation in Mato Grosso and Pará".
- Goldemberg (2008) states that although there are concerns about the sugarcane cultivation area expansion generating an indirect pressure pushing cattle into the Amazonia and leading to further deforestation in that area, there is no direct evidence for that.
- Other studies have been more nuanced, arguing that additional demand for sugarcane, along with many other factors such as demand for soybean and urbanisation, contributed to increasing the prices of land in the Centre-South region, which made it unprofitable or uninteresting for ranching activities (Barona et al., 2010; Schlesinger, 2008 and 2010). Furthermore, government investment in the Amazonian region, especially in infrastructure building, opened up a new region where land prices were low and claiming of the land relatively easy (Walker et al., 2009). It is, however, impossible to separate out how much each factor contributed to the displacement of pasture area and made it possible for pasture to expand in the North.

Based on these observations, two sets of scenarios were analysed in this study, which differed in terms of whether or not it was assumed that there was a causal link between sugarcane expansion and pasture displacement. Although the reality is likely to lie somewhere between these two polarised views, by exploring the ILUC impacts in these two scenarios, it is possible to gain an understanding of the difference this effect has on the ILUC impact calculation:

1. Sugarcane expansion *is not* causing pasture displacement. This means that pasture displacement would happen anyway, regardless of the additional sugarcane demand. In the baseline, the land freed up by pasture displacement would be replaced by other crops. However, in the biofuels projection, part of the land freed up by pasture displacement would be replaced by sugarcane. Thus, the actual type of land use change that happens in the biofuel projection is other crops (assumed to be annual crops) being replaced by sugarcane.

   Furthermore, the other crops no longer grown on the displaced pasture land will need to be grown somewhere else, leading to another "knock-on effect".
2. Sugarcane expansion *is* causing pasture displacement. This means that additional expansion of sugarcane in the Centre-South region will lead to additional displacement of pasture land.

Thus the actual land use change taking place is the conversion of grassland (pasture) to sugarcane cultivation area.

Again the cattle produced on the displaced pasture land will have to be produced somewhere else. We have assumed that all the cattle would be displaced to the Northern region of Brazil. The exact amount of land by which pasture expands in the North due to this displacement depends on assumptions around the cattle stock rate in the Centre-South region and in the North region and the assumed pasture intensification rate to 2020 (see next section).

Pasture intensification rate. Several parameters affect the land used for pasture in Brazil. Apart from the shift from one region to another, the intensity (or yield) of pasture land is also increasing, with rates depending on the region. Figure 39 presents the evolution of stock rates between 1970 and 2006 for Brazil as a whole and for two different regions of Brazil.

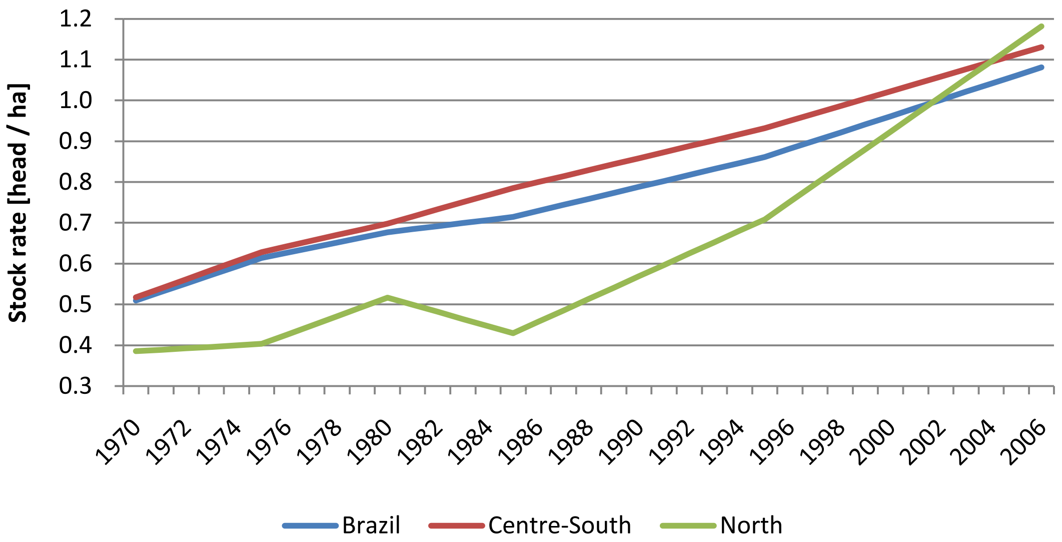

Figure 39. Historical evolution of pasture stock rates in Brazil as a whole and in the Centre-South and the North region.

Source: data from the Instituto Brasileiro de Geografia e Estatistica (IBGE) Agricultural Census, compiled by Instituto de Estudos do Comércio e Negociações Internacionais (ICONE) (Nassar, 2010b).

Historically, the North of Brazil has had very low pasture stock rates. While stocking rates in Brazil as a whole and the Centre-South in particular were slowly but almost constantly increasing, stocking rates in the North only started to increase in the mid 1980's. Since then, stocking rates have increased more rapidly than in other regions and currently pasture is more intensive there than in the Centre-South.

This means that currently one hectare of pasture converted to crops in the Centre-South region may cause less than one hectare of pasture expansion in the North (Nassar, 2010a). Our inherent assumption in this analysis is that higher pasture intensification will result in less deforestation. However, this is a simplification, as greater pasture intensification does not automatically ensure that there will be less deforestation. Pasture expansion is a contributing factor to deforestation but it is not the only cause. In other words, whilst pasture intensification may be a necessary factor for slowing the rate of deforestation, it is unlikely to be sufficient to halt it.

Uncertainties are high around the future of pasture intensification rates. These depend on land prices, government subsidies, meat prices, etc. Furthermore, as these rates determine the amount of land that will expand onto the Amazonian forest, they influence the ILUC impacts calculated in this study. This study looks at four different scenarios to understand the link between pasture stock rates and ILUC factor of sugarcane:

1. Extrapolation of historical rates of increase in stock rate. The CAGR of stock rate in the Centre-South region from 1970 to 2006 was 2.2%, whereas in the North region, the CAGR was 3.2%. This leads to stock rate projections in the Centre-South in 2020 of 1.53 head / ha compared with 1.83 head / ha in North. This means that 1 ha pasture displaced in the Centre-South region leads to 0.84 ha pasture expansion in the North region.
2. Above baseline pasture intensification in the North leading to no pasture expansion. For 1 ha of displaced pasture in the Centre-South region to lead to 0 ha of pasture expansion in the North region, the stock rate in the North would have to increase to 1.9 or 2.0 head / ha. This stock rate depends on assumptions such as the number of cattle heads in the North region in the baseline scenario and the number of cattle heads displaced from the Centre-South region. Annex 8 provides more details on the assumptions behind this calculation.
3. Low pasture intensification in the North leading to the same stock rate for Centre-South and North in 2020. This means that 1 hectare of displaced pasture in the Centre-South region leads to 1 hectare of pasture expansion in the North.
4. Very low pasture intensification in the North. This would lead to 1 hectare of displaced pasture leading to more than 1 hectare pasture expansion in the North. In this scenario, a stock rate of 1.3 head / ha in the North vs. 1.5 head / ha in the Centre-South was assumed.

To determine the type of land onto which pasture expands in the North region, Winrock International data was used (US EPA, 2010). Table 63 presents this data.

Table 63. Types of land use change and emission factors associated with them for pasture in the North region of Brazil.
Based on Winrock data for the US EPA (2010).

Type of land	Share *(lower deforestation rate scenario)*	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	54% *(27%)*	92.6 t CO_2e / ha	± 98 t CO_2e / ha
Grassland	8% *(22%)*	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha
Mixed	15% *(15%)*	24.6 t CO_2e / ha	± 25 t CO_2e / ha
Savannah	20% *(34%)*	8.72 t CO_2e / ha	± 4.2 t CO_2e / ha
Shrub land	2% *(2%)*	26.2 t CO_2e / ha	± 4.2 t CO_2e / ha
Wetland	1% *(1%)*	13.1 t CO_2e / ha	± 4.2 t CO_2e / ha
Barren	0% *(0%)*	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

GHG emissions associated with pasture intensification. It is important to consider whether there will be any other change in emissions resulting from pasture intensification, other than the avoided LUC assumed here (although, as discussed above, even this is not a consequence that may be taken for granted). For example, pasture intensification may have a positive impact on emissions due to lower methane emissions from cattle[34], improved carbon sequestration in soils due to better pasture management (Smeraldi, 2010) or generally more efficient cattle management. Alternatively, pasture intensification may have a negative impact on emissions, for example, if more carbon intensive practices are employed in ranching or if the increased profits and productivity brought about by intensification result in an increased demand, and causes expansion of pasture land onto new frontiers[35]. Adding to these complexities is the uncertainty around whether there will be the necessary financial support, education and regulatory framework to enable pasture land to be intensified in an environmentally sustainable way (Juleff, 2010).

Those studying these effects point out that these issues are so complex that it may not be possible to accurately estimate emissions per kg meat. In addition, it becomes difficult to know where to draw the boundaries for the analysis. For example, with pasture intensification, there will be some small scale ranchers that are not able to intensify and as a result, may migrate to cities and potentially lead a more or less carbon intensive lifestyle as a result (Villarreal, 2010).

Given these complexities around understanding whether the different levels of pasture intensification in the baseline and biofuel projection will result in more or less GHG emissions, it has not been possible to quantify the magnitude of this effect in this study. However, we would like to flag this as an important area for further investigation and study, and highlight the importance of improving our collective understanding of the impacts in both GHG and social terms of pasture intensification.[36]

Deforestation rate. Deforestation is a major environmental problem in Brazil – with a large part of the Amazon forest being cut-down every year. In this study, Winrock International (US EPA, 2010) data on the type of land being converted to agricultural land between 2000 and 2007 was used, i.e. historical data.

Several reasons for deforestation have been put forward in recent years, such as the increasing urbanisation of the Amazon region or of other Brazilian regions, industrial logging, pasture expansion in the Amazon region (Nassar, 2010a), which itself may be due to increased international demand for meat and other agricultural products (de Fries et al., 2010), etc. It is probably a combination of these underlying mechanisms that have pushed deforestation.

Recently, some evidence has been provided that the deforestation rate has been decreasing in Brazil (UNICA, 2008b; Carrington, 2010). However, there is ongoing discussion about whether this

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[34] Intensification has been suggested as a means of reducing methane emissions from cows – methane is a by-product of enteric fermentation of carbohydrates (Muller and Bartsch, 1999) and intensification results in shorter cattle lifespan (whilst maintaining the same level of meat production) and more protein and less carbohydrate in the feed, both of which result in less methane production per unit of meat produced.

[35] It should be noted that such further expansion onto new frontiers would also be dependent on expanded industrial slaughtering capacity in frontier areas and no or little restriction from public bodies (Smeraldi, 2010).

[36] This is an area of increasing activity, for example, an international workshop was held in Sao Paulo in 2009 about solutions to deforestation and greenhouse gas emissions caused by cattle expansion. More information can about this workshop can be found here: http://www.bioenergywiki.net/International_Workshop_on_Solutions_to_Deforestation_and_Greenhouse_Gas_Emissions_Caused_by_Cattle_Expansion#Meeting_Aims_and_Activities

decrease will be sustained in the long term and whether it is just a reflection of the global economic downturn (Butler, 2010).

To explore the influence of deforestation rates in Brazil on the total ILUC impact, two different deforestation rates have been used in different scenarios, with all other parameters kept constant. The first rate was a continuation of the historical rate (i.e. use of Winrock International data). The second rate was set at half of the historical deforestation rate. The resulting land conversion data can be seen in Table 61 and Table 63 (number in italics and parenthesis in the column, “Share”).

8.4.5.2 Land use change in the rest of South America

The rest of South America is assumed to supply 29 to 85 million tonnes of additional sugarcane. As for Brazil, the area under sugarcane cultivation is decreasing in the baseline but not in the biofuel projection, thus two land use effects can be identified:

1. the avoided reversion of 4.6 thousand ha due to the no longer decreasing sugarcane cultivation area;
2. the actual expansion of between 306 and 787 thousand ha (depending on the scenario considered) of sugarcane cultivation area.

Table 64 below shows the data used to assess the GHG consequences associated with the avoided reversion in the rest of South America. The following paragraphs then details the GHG consequences associated with actual sugarcane cultivation expansions.

Table 64. Type of land that would have replaced sugarcane cultivation in South America (excluding Brazil) and the reversion factors associated with them.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Reversion data	
		30 year reversion factors	95% confidence interval
Forest	48%	57.3 t CO_2e / ha	± 69 t CO_2e / ha
Grassland	14%	-6.37 t CO_2e / ha	± 16 t CO_2e / ha
Mixed	8%	19.4 t CO_2e / ha	± 27 t CO_2e / ha
Savannah	16%	-3.01 t CO_2e / ha	± 17 t CO_2e / ha
Shrub land	14%	4.99 t CO_2e / ha	± 20 t CO_2e / ha

Type of land expanded onto. As for the previous cases, sugarcane cultivation area expansion in the rest of South America will lead to a knock-on effect, and Winrock International (US EPA, 2010) data is used. Table 65 below summarises this data.

Table 65. Types of land use change and emission factors associated with them for perennial crops in South America (excluding Brazil).
Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	**95% confidence interval**
Forest	19%	73.4 t CO_2e / ha	± 94 t CO_2e / ha
Grassland	19%	-4.46 t CO_2e / ha	± 8.1 t CO_2e / ha
Mixed	24%	15.2 t CO_2e / ha	± 28 t CO_2e / ha
Savannah	24%	1.57 t CO_2e / ha	± 12 t CO_2e / ha
Shrub land	11%	13.6 t CO_2e / ha	± 21 t CO_2e / ha
Wetland	1%	4.59 t CO_2e / ha	± 14 t CO_2e / ha
Barren	2%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.4.6 Land use change from Southern African supply

Southern Africa is assumed to supply between 1 and 7 million tonnes of additional sugarcane. The land use impact of such an increase in production was determined in section 8.3.2.7 and corresponds to an expansion of the planted sugarcane area of between 20 and 101 thousand hectares, depending on the assumption about yield increases.

Type of land expanded onto As for the previous cases, sugarcane cultivation area expansion in Southern Africa will lead to a knock-on effect. Again, the knock-on effect is assumed to end in Southern Africa, and Winrock International (US EPA, 2010) data is used. Table 66 below summarises this data.

Table 66. Types of land use change and emission factors associated with them for perennial crops in Southern Africa.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	**95% confidence interval**
Forest	10%	32.3 t CO_2e / ha	± 67 t CO_2e / ha
Grassland	20%	-5.54 t CO_2e / ha	± 9.2 t CO_2e / ha
Mixed	14%	3.76 t CO_2e / ha	± 19 t CO_2e / ha
Savannah	36%	-0.37 t CO_2e / ha	± 10 t CO_2e / ha
Shrub land	14%	9.96 t CO_2e / ha	± 13 t CO_2e / ha
Wetland	0%	2.21 t CO_2e / ha	± 11 t CO_2e / ha
Barren	6%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.4.7 Land use change from Thai supply

Thailand is assumed to supply between 23 million tonnes of additional sugarcane. The land use impact of such an increase in production was determined in section 8.3.2.8 and corresponds to an expansion of the planted sugarcane area of between 227 and 312 thousand hectares, depending on the assumption about yield increases.

Type of land expanded onto As for the previous cases, sugarcane cultivation area expansion in Thailand will lead to a knock-on effect. Again, the knock-on effect is assumed to end in Thailand, and Winrock International (US EPA, 2010) data is used. Table 67 below summarises this data.

Table 67. Types of land use change and emission factors associated with them for perennial crops in Thailand.
Based on Winrock data for the US EPA (2010).

Type of land	Share	Conversion data	
		30 year conversion factors	95% confidence interval
Forest	12%	97.0 t CO_2e / ha	± 92 t CO_2e / ha
Grassland	10%	-4.43 t CO_2e / ha	± 14 t CO_2e / ha
Mixed	48%	21.2 t CO_2e / ha	± 26 t CO_2e / ha
Savannah	23%	1.62 t CO_2e / ha	± 14 t CO_2e / ha
Shrub land	5%	13.7 t CO_2e / ha	± 14 t CO_2e / ha
Wetland	1%	4.65 t CO_2e / ha	± 14 t CO_2e / ha
Barren	0%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.4.8 Land use change from US supply

The United States is assumed to supply up to 23 million tonnes of additional sugarcane (excluding the scenario where the US is not supplying any of the additional sugarcane). The land use impact of such an increase in production was determined in section 8.3.2.9 and corresponds to an expansion of the planted sugarcane area of up to 207 thousand hectares, depending on the assumption about yield increases.

Type of land expanded onto As for the previous cases, sugarcane cultivation area expansion in the United States will lead to a knock-on effect. Again, the knock-on effect is assumed to end in the US, and Winrock International (US EPA, 2010) data is used. Table 68 below summarises this data.

Table 68. Types of land use change and emission factors associated with them for perennial crops in the United States.
Based on Winrock data for the US EPA (2010).

Type of land	**Share**	**Conversion data**	
		30 year conversion factors	**95% confidence interval**
Forest	6%	107 t CO_2e / ha	± 72 t CO_2e / ha
Grassland	36%	-7.00 t CO_2e / ha	± 5.3 t CO_2e / ha
Mixed	24%	19.4 t CO_2e / ha	± 21 t CO_2e / ha
Savannah	18%	-3.01 t CO_2e / ha	± 7.5 t CO_2e / ha
Shrub land	14%	4.99 t CO_2e / ha	± 13 t CO_2e / ha
Wetland	1%	-1.01 t CO_2e / ha	± 8.7 t CO_2e / ha
Barren	1%	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

8.5 Scenario results

Figure 40 presents the ILUC factors for all fourteen sugarcane bioethanol scenarios described. For all the scenarios, the ILUC factor is positive and varies from ~8 to ~27 g CO_2e / MJ. Compared with some of the other feedstocks reviewed in this report, the variation between the ILUC impacts calculated for the different scenarios is not particularly large, although it is significant.

The variation in the ILUC impacts would be more extreme if “best” and “worst” scenarios are considered. This was not explored in detail, as the focus of this study was to develop more realistic scenarios. However, a quick calculation can be made to explore what the highest and lowest ILUC factors would be if one looked at scenarios where all the parameters increasing / decreasing the ILUC factors were combined. For example, the best scenario would be based on scenario 11 but with a lower than historical deforestation rate in Brazil. The quick estimate shows that the lowest ILUC factor would be around 2 g CO_2e / MJ while the worst could be around 33 g CO_2e / MJ. The impacts could potentially be worse, if one calculated for example an ILUC factor with 100% expansion of crop and pasture onto forest.

As with the other fuel chains, no central scenario is provided, as it was felt that it would be too difficult to assign probabilities to each of these scenarios. Whether one specific scenario is closer than the others to the future will depend on not only biofuel mandates and support but also on other policies affecting deforestation, pasture, crop yields, etc.

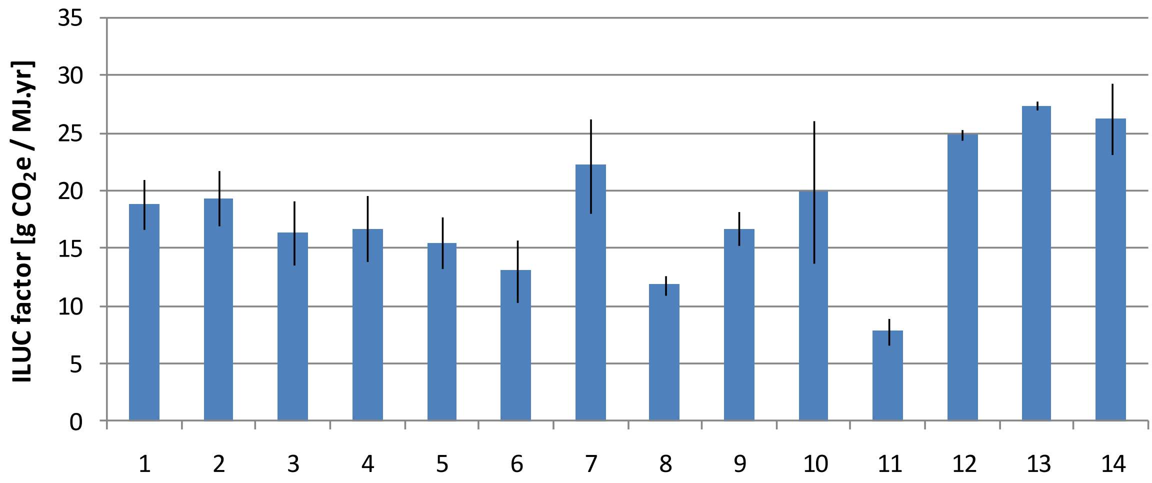

Figure 40. Indirect land use change impacts for the different scenarios modelled for sugarcane bioethanol.

Parameters that lower the ILUC impact of sugarcane bioethanol are, for example, the displacement of crop cultivation area to Argentina rather than Brazil, high rates of pasture intensification in North and Centre-South region and lower deforestation rates in Brazil. Parameters that increase the ILUC impact are assuming no above-baseline yield increases and low pasture intensification rate in the North region. The following points provide a more detailed explanation for the variation in the ILUC impacts shown in Figure 40:

- Including a small amount of domestic sugarcane production in the US and a small level of export from Southern Africa to the EU slightly lowers the ILUC factor by ~0.5 g CO_2e / MJ compared to assuming no production in the US and no export in Southern Africa (compare scenario 1 and 2). This is due to the fact that additional production of sugarcane in Southern Africa and the US leads to the conversion of land with lower carbon stocks than the land converted in Brazil. As such, these assumptions had very little overall effect on the ILUC impacts calculated.

- By assuming that the soy cultivation area displaced by sugarcane expansion in Brazil is displaced to Argentina, lower ILUC impacts (~3 g CO_2e / MJ lower – compare scenario 2 and 3) are calculated than if the soy is assumed to be grown elsewhere in Brazil. This is due to the assumptions about the levels of expansion onto different types of land in different countries, and also the lower levels of carbon stock in forests in Argentina compared with Brazil.

- Contrary to what one might expect, increasing the demand for sugarcane ethanol in the EU and US in the biofuel projection actually resulted in a lower ILUC factor (a reduction of ~2 g CO_2e / MJ – cf. scenario 1 and 4). The reason for this is that in the higher demand scenarios, a greater percentage of the sugarcane ethanol is assumed to come from Brazil, where yields are higher and the carbon stocks of the land are slightly lower than other key producing regions (e.g. Central America and the Caribbean). Thus the ILUC impacts (when calculated on a per MJ bioethanol basis) of Brazilian sugarcane ethanol (as modelled here) are lower than in those other countries.

- Assuming a lower deforestation rate in Brazil reduces the ILUC impacts by about ~3 g CO_2e / MJ bioethanol (see scenarios 1 and 5 or 4 and 6). GHG emissions from deforestation represent about 40% of the various ILUC factors presented (when considering historical deforestation rates). Although a 3 g reduction in the ILUC factor due to deforestation may seem small in absolute terms, it should be remembered that this represents ~20% of the total ILUC factor.

- Scenario 6 combines a situation with higher ethanol demand in the EU and US and a lower deforestation rate, which together reduce the ILUC impacts relative to scenario 1 by ~5.7 g CO_2e / MJ.
- Whether or not it is assumed that expansion of the sugarcane cultivation area causes pasture land displacement can have a significant impact on the ILUC factor. The actual impact depends on how pasture intensification evolves in the region it is being displaced to. If pasture intensification in the region continues at recent historic rates (i.e. more intensified than in the Centre-South region where it is assumed to be displaced from), the modelled ILUC impacts are lower (scenario 7) than if identical rates of pasture intensification occurs in both regions (scenario 12) or lower rates of pasture intensification in the North than Centre South (scenario 13), but higher than if an above baseline rate of pasture intensification is assumed in the North (scenario 8).
- The final parameter assessed in scenario 14 is the effect of there being no biofuel demand induced yield increase. The effect of doing so increased the ILUC impacts by ~7.5 g CO_2e / MJ.

Although several of the scenarios combine these different effects, not all possible combinations are shown. The breakdown of ILUC factors presented in Annex 7 can be used to estimate the impact of other scenarios not shown here.

Figure 41 shows the contribution of each market response to the overall ILUC factor calculated for scenario 10. As described in earlier sections of this chapter, the only modelled market responses to the increased demand for sugarcane ethanol is expansion in sugarcane area. For the breakdown of the ILUC impacts of other scenarios, please refer to Annex 7.

As can be seen from the diagram, the ILUC impacts result entirely from expansion of sugarcane onto land in different countries, with the largest impacts caused by expansion of sugarcane in Brazil. This is because our model projects Brazil as the country where the most expansion of sugarcane production will take place. There are two ILUC “credits” (shown in green in the diagram) which represent sugarcane land expanding onto pasture land or other crop land. The reason that this results in an ILUC credit is because the sugarcane crop is assumed to have a higher carbon stock, on average, than the pasture (grass) land or the other (assumed annual) crops grown on the land (e.g. soybeans). It should be noted that although this results in an ILUC “credit”, the displacement of these land uses to elsewhere results in a knock-on ILUC “debit” as the pasture and crop land displaces higher carbon stocks in the place they are displaced to. The combination of these effects leads to an ILUC debit for sugarcane expansion in Brazil.

For other regions where additional demand for sugarcane leads to ILUC this level of detail is not provided, and only the overall debit due to additional sugarcane production is shown.

It is also interesting to note from the error bars in the diagram that the largest degree of uncertainty around the carbon stocks data is for the large amount of carbon stock assumed to be lost due to expansion of pasture land in the Amazon.

As discussed earlier in this chapter, the link between sugarcane cultivation expansion, pasture displacement and/or intensification and deforestation in Brazil is an area of ongoing discussion and research. Based on our analysis, deforestation was identified as a strong contributor to the total ILUC impact of sugarcane bioethanol. For example, in scenario 1, 40% of the GHG emissions are due to deforestation. However, it is a simplification to describe sugarcane expansion in the Centre-South region of Brazil as the sole cause of deforestation in the North, and attribute all emissions to the sugarcane expansion; there are other drivers for deforestation such as timber and increasing demands for land ownership.

Lowering the deforestation rate in Brazil is high on the political agenda worldwide, and this is not necessarily linked to any biofuel policies. This is why we have modelled separately lower deforestation rates and other scenarios such as pasture displacement and intensification. There are those who do see a clear link between deforestation and pasture displacement and intensification. The World Bank Group (2010) identified increasing pasture productivity as the main way of lowering the deforestation rate, as pastures have a greater potential to increase productivity than other crops in Brazil. Further work could be done to better understand exactly the contribution that sugarcane expansion has on deforestation and pasture displacement.

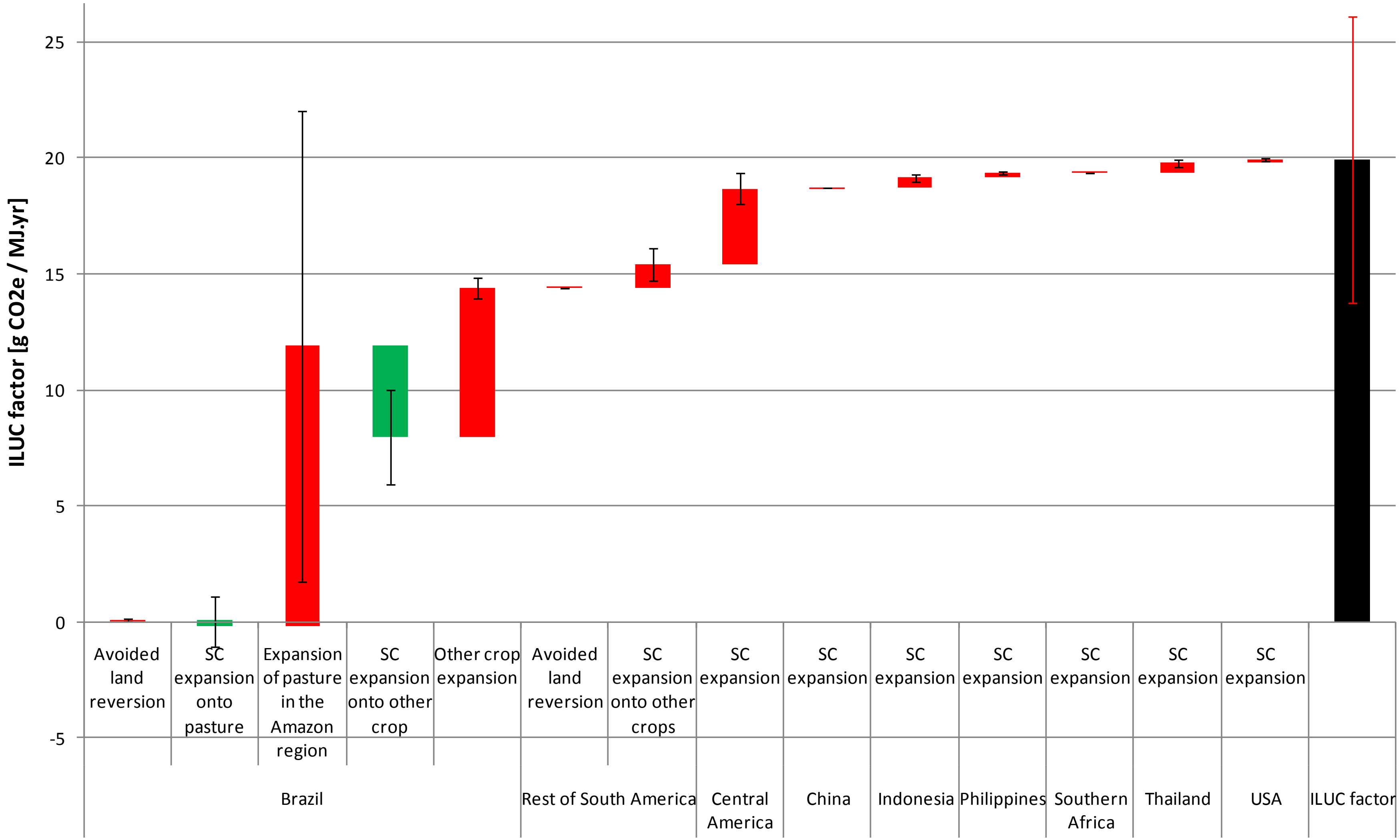

Figure 41. Waterfall diagram showing the contribution of each market response to the overall ILUC factor for scenario 10.

9 Actions to mitigate the magnitude of the ILUC impact

9.1 The need for ILUC mitigation

As shown in the previous chapters, GHG emissions from ILUC can be significant, reducing the potential for GHG emission reductions of biofuels, or even off-setting them completely. Thus, in order for biofuels to be a viable GHG emission reduction technology, the risk of biofuels causing ILUC needs to be mitigated.

Two main policy mechanisms for ILUC risk mitigation commonly discussed today:

- The ILUC factors approach. This approach involves adding an ILUC factor to the direct GHG emissions of a biofuel production chain. Such an approach would mitigate ILUC if policy favoured those biofuels with lower ILUC impacts. ILUC factors have been applied in biofuel policies in the United States such as the US RFS 2 or the Californian Low Carbon Fuel Standard (LCFS).
- Action-based approach. This approach consists of implementing actions that mitigate the risk of ILUC. Such actions can be taken at different levels, from the project or farm-level to more regional, national or supra-national levels. Examples include intensification of agricultural production or forest protection policies.

There are difficulties and drawbacks to each of these policy mechanisms, as described in the following sections. However, biofuels are not the only products which can cause ILUC. In fact, any product can cause ILUC, from fibre and food/feed products to fossil fuels. ILUC is a system problem that concerns the agricultural and forestry systems more broadly. While biofuels may add to the ILUC pressure, their overall impact in relation to LUC is likely to be relatively small in the short term compared to the demand for other agricultural products. An ILUC mitigation policy has more chance of being effective the more sectors it encompasses.

9.1.1 ILUC factors

As a scientific way of improving our understanding of the scale and type of ILUC impacts, ILUC factors are increasingly discussed and modelled. However, they present a number of difficulties as a policy tool aimed at ILUC mitigation.

ILUC factors present a static picture of the ILUC effects of biofuels. In contrast, ILUC impacts are dynamic and can change significantly as a result of changes over time in the agricultural system and land use, and demand for different agricultural products.

Also, because of the dynamic nature of ILUC factors, using them as a way of influencing demand for different biofuels will affect the ILUC factors themselves, as ILUC factors are dependent on the amount of additional demand for biofuel feedstocks. Thus, significant shifts in demand for lower ILUC factor biofuels could increase their ILUC factor.

ILUC factors are uncertain due to uncertainties associated with the estimation of the amount of LUC that occurs as a result of demand for biofuel feedstocks, and as a result of uncertainties associated with magnitude of carbon stock lost on that land (both from removal and avoided reversion).

Therefore a number of questions arises, for example, how often would the ILUC factor need to be updated, to remain relatively accurate? Would the ILUC factor only remain "accurate" as long as the amount of a certain type of biofuel used didn't go above a threshold?

9.2 ILUC mitigation actions

Mitigation actions provide a means of controlling ILUC effects, and recent reports have focused on these (Ecometrica, 2010; Ecofys and Winrock International, 2010). These reports provide extensive information on actions at different geographical levels and their associated indicators. Also, the causal-descriptive approach presented in this report highlights specific actions that have the potential to significantly reduce the ILUC impacts identified, and these actions reflect those discussed in the above-mentioned reports. Interconnectedness of the food, feed, fibre and biofuel markets means that the impact of ILUC depends on practices in supplying the different markets and so does the mitigation of ILUC impacts.

The net extent of land area expansion depends on several factors associated with crop production chain efficiencies, biofuel production chain efficiencies, and use of co-products. Actions to mitigate ILUC can be aimed at:

1. Controlling the type and extent of land use change. This includes actions such as protecting high carbon stock areas, directing biofuel feedstock cultivation to unused degraded land areas, or better land-use planning.
2. Controlling the factors that affect the extent of land use change. This includes actions such as improving crop yields, improving biofuel production efficiency or ensuring biofuel co-product are used to displace land-based products.

Through the causal-descriptive approach taken for calculating the ILUC impacts of five different biofuel chains, we have identified a series of mitigation actions, which will be discussed in the two next subsections. Section 9.2.3 then discusses possible indicators for monitoring the effectiveness of the ILUC mitigation actions.

9.2.1 Actions to control the type and extent of land use change

Protect high carbon stock areas (such as forest, peatland, etc.). Protecting high carbon stock land prevents the conversion of this land into agricultural use and thus avoids the associated GHG emissions. Such an action may not affect the extent to which land use change would happen, but by protecting high carbon stock areas, it leads to lower GHG emissions from the LUC.

The impact of such actions on the ILUC factors can be observed through the ILUC scenarios that assumed lower than historical deforestation rates in Indonesia and Malaysia or Argentina and Brazil, such as case 6 for palm biodiesel (low deforestation rates in Indonesia and Malaysia and low peatland conversion rate – see chapter 4) or case 4 for oilseed rape biodiesel (low deforestation rate in Indonesia and Malaysia – see chapter 5). In both cases, these assumptions lead to ILUC factors in the lower end of the calculated range.

The example of wheat (chapter 7) is slightly counter-intuitive. Wheat ILUC scenario 7 assumed lower deforestation rates in Brazil and Argentina. However, as these deforestation rates only have an impact on the credit received due to the bioethanol co-product displacing soybean meal (i.e. decrease the credit), this assumption actually leads to a higher, but still negative, ILUC factor.

Use lower carbon stock areas for biofuel feedstock cultivation (taking into consideration other environmental impacts) such as degraded or abandoned land. This is, for example, what the European Renewable Energy Directive (EC, 2009a) is aiming at with the provision that a credit of 29 g CO_2e / MJ biofuel can be given to biofuels produced on degraded land. However this could only really be argued to be mitigating ILUC if it could be shown that the biofuel feedstock grown on that low carbon stock land does not displace other crops or activities that would otherwise expand onto that land (and instead expand onto higher carbon stock land). This again highlights the importance of applying mitigation actions across all land-based activities.

Special attention should be given in this case to foregone sequestration of the carbon that would be stored by the land that is no longer abandoned.

Implement land use zoning based on current land use. This action consists of deploying strategies to 'zone' land according to land type and use and defines where and what land use changes are permitted and where existing land uses (e.g. primary forest) are to be protected.

Ensure all converted land remains productive for a long time period (i.e. managing the land so that it does not have to be abandoned after e.g. 30 years cultivation). It is important to ensure that, where possible, no other land has to be converted as a result of the degradation and abandonment of converted land.

Land use zoning and biofuel policies need to be long term policies, rather than ways of quickly fixing a pollution or climate change issue, as the benefits are clearly greater if maintained over the long term.

9.2.2 Actions to control the factors that affect the extent of land use change

Increase agricultural yields. Increased yields and/or agricultural intensification will increase the overall productivity of existing agricultural land: less land will be needed to satisfy the demand for land-based products. Thus higher yields reduce the extent to which land use change will happen.

The impact of yield improvement on the size of the ILUC factor has been shown through several ILUC scenarios. For palm, the comparison of case 1 and 4 (see chapter 4) shows that considering a 16% above baseline improvement in yields can lead to a 16% decrease in the ILUC factor. Wheat ILUC scenarios 3 and 4 (see chapter 7) also show that higher yield increases lead to less ILUC. Similarly, in sugarcane scenario 14, where no biofuel demand induced yield is assumed, the ILUC impacts are ~40% larger (equivalent to an increase of ~7 g CO_2e / MJ).

During the course of the project, there have been extensive discussions on how yield increases would be achieved and whether GHG emissions from these techniques should be considered in the ILUC factor. Several stakeholders have pointed to a list of yield improvement techniques that include, among others:

- Increased nitrogen fertiliser application;
- Better timing of nitrogen fertiliser application;
- Better weed management practices;
- Switch in varieties grown;
- Investment in agricultural research and development (R&D).

Some of these options have associated indirect GHG emissions. It has not been possible to include in this study the indirect GHG emissions associated with yield improvement. This issue has been discussed in section 3.3.4.2. However, these should be looked at further if considering yield improvement as an ILUC mitigation action.

For example, a policy supporting increased yield can lead to increased nitrogen fertiliser application per hectare. But, greater fertiliser application could lead to indirect GHG emissions if the increase in fertiliser leads to an increase in fertiliser per tonne of crop produced across the planting of the crop for biofuel and other purposes.

Improve supply chain efficiency. It was discussed in section 3.3.1 that we would not consider improvements in supply chain efficiency when calculating ILUC factors. However, it is important to address this possibility in this section on mitigation actions, as such an action could significantly affect the extent to which ILUC occurs. The less feedstock that is lost in the supply chain, from agricultural production to end market, the less will need to be produced, thus reducing the pressure on land.

Several actions can improve the supply chain efficiency, including lowering losses (through better pest management practices for example) or reducing supply chain wastage (through better storage and transport for example).

Use co-products from the production of biofuels to replace land-based products (such as animal feed). In this way, co-products will help to lower the increase in production of animal feed and avoid land use change. It was, for example, shown in the previous chapters (see chapter 5 on biodiesel from oilseed rape and chapter 7 on bioethanol from wheat) that such uses give higher GHG emission savings than co-firing the co-products for heat and power generation.

Integrate crop and livestock systems to increase land productivity. The rationale behind this approach is that, for example, sugarcane production can be introduced onto ranch land but existing levels of milk or beef production are maintained on the ranch land as well, through supplementing the animals' feed with sugarcane residues (thus the animals require less grazing area for the same level of productive output). This "land displacement" mitigation action is discussed in more detail in a recent report by Ecofys and Winrock International (2009).

9.2.3 Indicators

Indicators are aimed at monitoring the extent to which ILUC is effectively mitigated. Ecometrica (2010) identify two categories of indicators:

- Action-based indicators, that relate to specific actions. They show whether an action has effectively been undertaken. However they do not show whether ILUC has effectively been mitigated. For example, expenditure on agricultural research and development is an indicator related to the specific action of investing in agricultural R&D. However, it is not certain that, if the indicator shows increased investment then lower ILUC is achieved.
- Outcome-based indicators, that provide high-level information about the likelihood of ILUC occurring. Two main difficulties in using outcome-based indicators have been identified:
    - ILUC mitigation actions are not the only parameters that can influence an outcome-based indicator. For example, declining exports from a region (which points to increased likelihood of ILUC effects outside the region) can be a consequence of declining demand

for the exported commodities or competition from other regions and not necessarily reflect an increased demand for the commodity because of biofuels.

- Outcome-based indicators should be expressed relative to a baseline. If increasing yield is monitored, the important indicator is not whether yield has increased but whether it has compared to the baseline scenario where no additional demand for biofuel feedstock exists (i.e. the above baseline increase).

Table 69 provides an overview of ILUC mitigation actions identified in this report and examples of possible associated indicators.

Table 69. Overview of mitigation actions and possible mitigation indicators

Mitigation action	Associated outcome-based indicators	Associated action-based indicators
Protect high carbon stock areas	• Total land-based carbon stock in a certain region • Area of high-carbon-stock land converted to agricultural production • Forested area in a certain region • Peatland area in a certain region	• Total amount of protected area in a certain region • Public investment into area protection in a certain region
Use low carbon stock areas for biofuel feedstock cultivation	• Total land-based carbon stock in a certain region	• Policy incentives for the use of marginal, degraded or abandoned land
Impose land use zoning based on current land use	• Total land-based carbon stock in a certain region • Conversion of high-carbon-stock land to agricultural production	• Percentage of the total area covered by land use zoning policy
Ensuring converted land remains productive for a long time period	• Total amount of abandoned land	• Stable biofuels policy
Increase agricultural yields / intensification	• Yield improvements above baseline	• Introduction of N application techniques • Investment in agricultural R&D
Improve the supply chain efficiency	• Yield improvements across supply chain	• Infrastructure improvement • Expenditure on infrastructure development
Ensure the co-products from the production of biofuels are used to replace land-based products	• Share of co-products used in animal feed	• Policies directing use of biofuel co-products

Mitigation action	Associated outcome-based indicators	Associated action-based indicators
Integration of crop and livestock systems	• Baseline production levels of livestock maintained • Evidence of use of crop co-products in animal feed • Crops produced on prior ranch land • Average area of land owned not increasing	• Subsidies/support/training for ranchers/settlers to develop integrated farming systems

The mitigation actions and indicators identified are the most obvious and important in light of the causal-descriptive analysis carried out. A more extensive list of possible actions is provided by Ecofys and Winrock International (2010) and Ecometrica (2010). Ways of implementing these actions needs further consideration to help identify effective options for reducing ILUC impacts, and the barriers and difficulties associated with implementing them.

10 Discussion

10.1 The problem

Any additional demand beyond the current status quo for any crop (for any application) has the potential to lead to some non-agricultural land being brought into cultivation. The GHG emissions associated with bringing non-agricultural land into production vary depending on the type of land that is brought into production and the way that it is brought into production. Our analysis shows that depending on the type of product for which there is an additional demand and the assumptions made about the impact additional demand for that product has on other commodities, the amount of "new" land brought into cultivation and the associated GHG emissions may be very different. It also shows how indirect land use change depends strongly on the interdependencies within and between the agricultural system, land use and agricultural markets, and is not only relevant to biofuels but to all land–based products. By exploring the potential linkages between demand for a crop and global land use, it is possible to identify the factors and therefore practices that significantly affect the land use impact.

10.2 The value of the exercise

In order to review the value of this exercise, it is important to recapitulate on its purpose, which was to:

- help develop an understanding of what the causes and effects of ILUC are for specific biofuel chains;
- help understand which effects significantly contribute to the ILUC impact and therefore where mitigation measures could most usefully be focused;
- provide a transparent approach for engaging a wide range of stakeholders and capturing their insight;
- help inform other modelling approaches, e.g. econometric models.

It should be emphasised that although this work attempts to estimate the magnitude of "ILUC factors", the reason for doing so does not come from a view that ILUC factors would be an effective policy measure. Our view is that the cause-effect relationships can be used to begin to understand the kinds of actions, on a global level, that can help mitigate against this global and cross-sectoral problem, and provide a basis from which we can start to work out practical ways to reduce the ILUC impacts associated with biofuels and other global commodities.

ILUC factors provide a quantification of the impact but remain imprecise as a result of possible changes in the context and uncertainties in underlying assumptions and carbon stock figures. Potentially the implementation of an ILUC factor could change purchasing behaviour, alter volumes of crops grown and trade flows and therefore the ILUC factor itself. Also, as it is not possible to "observe" ILUC, it is not possible to scientifically test and validate the different approaches for modelling ILUC and developing ILUC factors. This makes it difficult to reach a consensus on the "right" assumptions for the different biofuel chains, especially when trying to estimate future impacts.

However, ILUC factors can be helpful in understanding the potential magnitude of impacts under certain situations, and therefore the risks posed by ILUC, and how they may change under different situations. Their derivation also helps understand the factors that affect them and hence how they can possibly be influenced.

10.2.1 Benefits of approach

This work helps to develop a bottom-up understanding of the causes and effects leading to ILUC for different biofuel chains. It has been possible to share views with and inform a broad range of stakeholders as well as integrate many of their views into this work. Where possible we have been able to use the latest data and market trends to predict the context governing ILUC impacts, and have also explored the sensitivity of the ILUC impacts to other likely future contexts.

In addition, we have provided both in the waterfall diagrams and in the appendices a breakdown of the ILUC factors into the different market responses that contribute to them. This has the benefit of enabling the reader to understand what the effect on the ILUC factor would be if alternative assumptions were made (not included within our scenarios), which could either increase or decrease the contribution of that specific component.

In the time available it has not been possible to build a perfect picture of all the causes and effects and model all the possible scenarios that could occur. However, this work does provide a framework for capturing these effects and a significant effort to model the key cause-effect relationships and scenarios. As always, with more time, the assumptions could clearly be refined further, including through additional discussions with stakeholders and analysis.

10.2.2 Limitations

There are certainly limitations with using this approach to calculate one single ILUC factor. It is not possible to estimate the exact ILUC impact associated with using a MJ of a biofuel because it is only based on projections of future demand and best estimates of system responses. The methodology also does not provide a precise range in which the ILUC impact sits, because there are many potential scenarios that have not been explored here. Specific limitations have also been highlighted throughout the study as they relate to specific assumptions or view-points made during the specific chain calculations.

As explained elsewhere in this report an "average" approach has been used for the type of land that is expanded onto in response to a demand for a particular amount of biofuel. This was viewed to be a robust approach, up to certain tipping point. For example, there will be a limit to the amount of palm biofuel that can be planted in Indonesia and Malaysia, in terms of land available in those countries. Beyond that tipping point, additional countries would need to be assumed to be growing greater amounts of the palm oil. However, we believe that, in terms of the quantities of biofuel being considered here, those tipping points are either not reached (as in the case of palm) or are dealt with through scenarios (as in the case of OSR and wheat).

10.3 Modelling outputs

The risk of ILUC associated with palm oil biodiesel could potentially be very large (impacts calculated in our scenarios range from 6-82 g CO_2e / MJ). The actual ILUC impact does not have to be large but there are a number of factors which imply a large impact. The historic precedent of palm expanding onto deforested land in the countries where it is grown means that there is a risk that this could continue, with a high ILUC factor risk associated with it. Another key factor leading to a high ILUC factor risk is the large areas of peatland in the key producing countries (peatland drainage has significant associated GHG emissions). For these risks to be significantly reduced, evidence is required to show that policy to protect high carbon stock land is acting effectively.

The risk of ILUC associated with rapeseed biodiesel is estimated to be lower than for palm biodiesel in this analysis, but is still large enough to be of concern (impacts calculated in our scenarios range from 15-35 g CO_2e / MJ). The scenarios in which the ILUC factors are larger are those in which more rapeseed is imported from outside Europe and where more conservative assumptions are made about the utility of the co-products and therefore the credits associated with them.

As described in detail in the soy chapter, the soybean biodiesel ILUC factors (calculated in this study to be in the range of 9-66 g CO_2e / MJ) are entirely dependent upon the palm ILUC factors. The fact that some palm oil is likely to substitute for soy oil going into biodiesel leads to a high ILUC factor risk. Also, this makes it very difficult to identify practices in soy cultivation that will reduce its ILUC impact. However, it also strongly emphasises the global nature of the ILUC problem and how a concerted effort is required globally to mitigate ILUC impacts for all crops for all sectors.

The risk of ILUC associated with wheat ethanol, as modelled here, is very low, and actually negative, relative to the other biofuels (calculated in this study to be in the range of -53 to -5 g CO_2e / MJ). The main reason why this is the case is due to the large credit that is given to the wheat bioethanol by assuming that wheat DDGS is used as an animal feed. However, even in a conservative scenario in which only 50% of the DDGS is assumed to be used as an animal feed, the ILUC risk is still negative. Decreasing European exports to North Africa could also lead to a higher ILUC factor, as this leads to wheat production in countries such as Australia and Canada that are assumed to have much lower yields than Europe.

The ILUC impacts calculated in the sugarcane ethanol scenarios explored in this study are in the range of 8 to 27 g CO_2e / MJ. These impacts are entirely caused by expansion of sugarcane onto new land in different countries. The scenarios in which the ILUC risks are greater are those in which current levels of deforestation are seen to continue, pasture intensification slows down in the Northern regions of Brazil (to equal or lower than the pasture intensification level in the Centre South), and there is no biofuel demand induced yield increase.

10.4 Managing risk

Several actions or practices can reduce the risk of ILUC. The key actions that would mitigate the risks identified through the methodology developed in this report are:

- protection of high carbon stock land;
- use of low carbon stock areas for biofuel feedstock cultivation;
- land use zoning, long term effective utilisation of an area of land;
- increase yields (above baseline increases);
- improve supply chain efficiency;
- ensure co-products from production of biofuels are used as replacement of land based products;
- integrated crop and livestock systems to make better use of land.

Some of these factors, e.g. above baseline yield improvements, can help mitigate ILUC risks even if just applied to the biofuel sector. However, it is less straightforward to be certain that other practices, e.g. use of low C stock areas for biofuel feedstock cultivation, are mitigating ILUC when applied only to the biofuel sector. ILUC risks could only be demonstrated to be mitigated if it was possible to demonstrate that biofuel feedstock grown on low C stock land does not displaced other crops that would otherwise expand onto that land (and expand instead onto higher C stock land).

In summary, there is a risk of indirect land use change associated with using biofuels. It is unclear as yet to what extent those risks can be mitigated through particular actions. As such, there is a need to both understand how risks can be mitigated effectively as well as continue to improve their quantification.

10.5 Further work

It has been highlighted throughout this report where further work is needed to improve the understanding of the magnitude of ILUC impacts using the methodology developed here. For example, we have identified where better datasets could be used to better represent the land use changes taking place in different countries. In addition to this, it is clear that more could be done to refine the assumptions developed in this work.

Interesting areas for future work include:

- additional scenarios to explore further cause and effect options;
- incorporating regional datasets on land use changes;
- improving understanding of the drivers of deforestation;
- further refinements of the assumptions through wider stakeholder engagement;
- application of the methodology to other fuel chains;
- a detailed comparison of the assumptions (and outputs) made here with those in other ILUC models based on other approaches (e.g. partial and general equilibrium models).

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Annex 1. Demand for feedstocks in the baseline and the biofuel projection

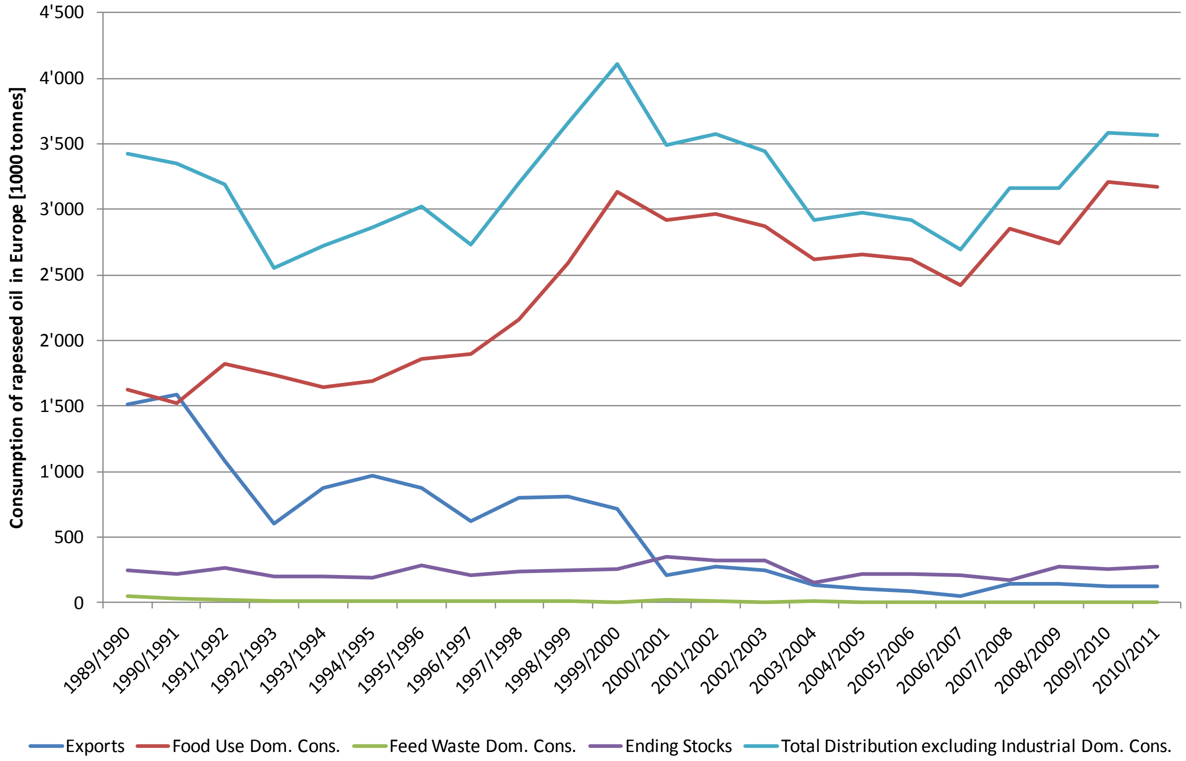

Figure 42. Evolution of rapeseed oil consumption in Europe from 1989/1990 to 2010/2011 for different end uses

Note: Dom. Cons. stands for domestic consumption. Source: USDA FAS (2010).

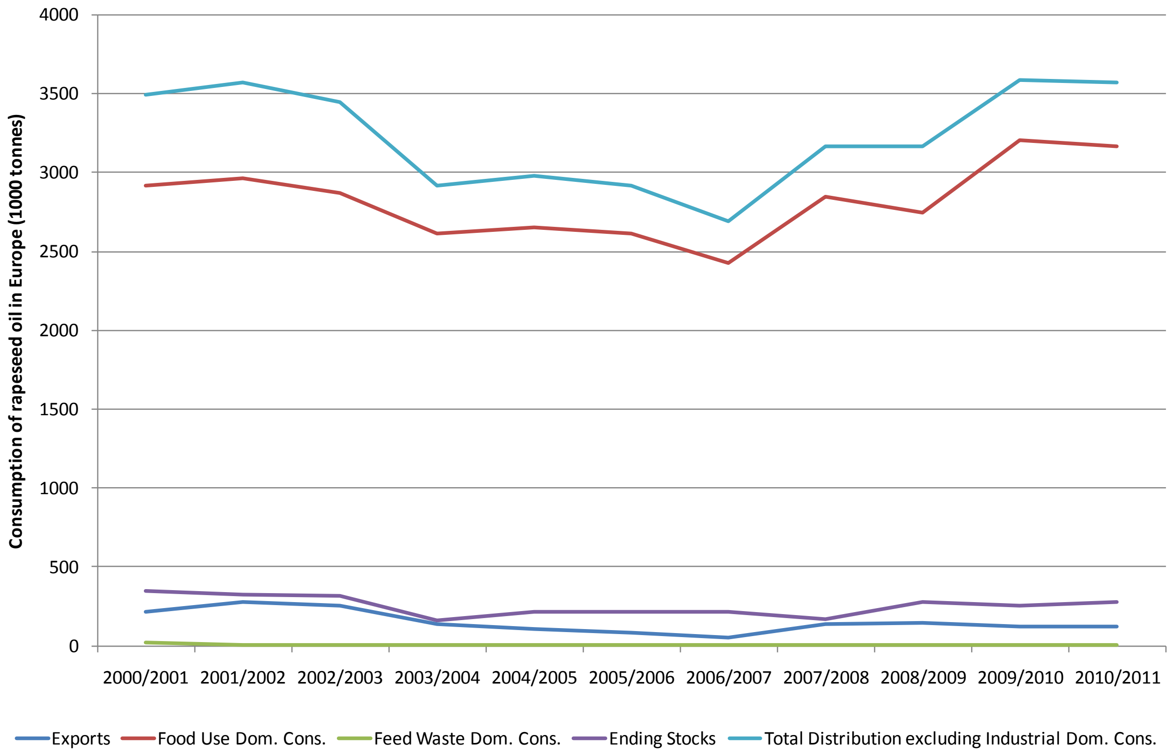

Figure 43. Evolution of rapeseed oil consumption in Europe from 2000/2001 to 2010/2011 for different end uses

Note: Dom. Cons. stands for domestic consumption. Source: USDA FAS (2010)

Annex 2. Emissions factors used for palm oil analysis

Table 70. Emissions from conversion of different land types in Indonesia to palm, assuming no conversion of peatland.
Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR Indonesia – No peatland	30 year emissions factor	30 year confidence interval	100 year emission factor	100 year confidence interval
Forest	154 t CO_2e / ha	± 95 t CO_2e / ha	157 t CO_2e / ha	± 95 t CO_2e / ha
Grassland	3.5 t CO_2e / ha	± 15 t CO_2e / ha	-27.7 t CO_2e / ha	± 9.7 t CO_2e / ha
Mixed	41.2 t CO_2e / ha	± 29 t CO_2e / ha	10 t CO_2e / ha	± 26 t CO_2e / ha
Savannah	9.5 t CO_2e / ha	± 16 t CO_2e / ha	-21.6 t CO_2e / ha	± 9.8 t CO_2e / ha
Shrub	21.6 t CO_2e / ha	± 18 t CO_2e / ha	-9.5 t CO_2e / ha	± 11 t CO_2e / ha
Wetland	12.6 t CO_2e / ha	± 17 t CO_2e / ha	-18.6 t CO_2e / ha	± 9.9 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

Table 71. Emissions from conversion of different land types in Indonesia to palm, assuming 5% of land expanded onto is peatland.
Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR Indonesia – Low peatland	30 year emissions factor	30 year confidence interval	100 year emission factor	100 year confidence interval
Forest	179 t CO_2e / ha	± 145 t CO_2e / ha	242 t CO_2e / ha	± 326 t CO_2e / ha
Grassland	29.0 t CO_2e / ha	± 85.6 t CO_2e / ha	57.4 t CO_2e / ha	± 286 t CO_2e / ha
Mixed	66.7 t CO_2e / ha	± 95.6 t CO_2e / ha	95.1 t CO_2e / ha	± 293 t CO_2e / ha
Savannah	35.0 t CO_2e / ha	± 85.6 t CO_2e / ha	63.4 t CO_2e / ha	± 286 t CO_2e / ha
Shrub	47.2 t CO_2e / ha	± 85.7 t CO_2e / ha	75.5 t CO_2e / ha	± 285 t CO_2e / ha
Wetland	38.1 t CO_2e / ha	± 85.6 t CO_2e / ha	66.4 t CO_2e / ha	± 285 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha

Table 72. Emissions from conversion of different land types in Indonesia to palm, assuming 33.3% of land expanded onto is peatland.
Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR **Indonesia – High peatland**	**30 year emissions factor**	**30 year confidence interval**	**100 year emission factor**	**100 year confidence interval**
Forest	307 t CO_2e / ha	± 94 t CO_2e / ha	670 t CO_2e / ha	± 95 t CO_2e / ha
Grassland	157 t CO_2e / ha	± 8.3 t CO_2e / ha	485 t CO_2e / ha	± 21 t CO_2e / ha
Mixed	195 t CO_2e / ha	± 25 t CO_2e / ha	523 t CO_2e / ha	± 30 t CO_2e / ha
Savannah	163 t CO_2e / ha	± 8.0 t CO_2e / ha	491 t CO_2e / ha	± 21 t CO_2e / ha
Shrub	176 t CO_2e / ha	± 7.7 t CO_2e / ha	504 t CO_2e / ha	± 20 t CO_2e / ha
Wetland	167 t CO_2e / ha	± 7.9 t CO_2e / ha	494 t CO_2e / ha	± 20 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

Table 73. Emissions from conversion of different land types in Malaysia to palm, assuming no conversion of peatland.
Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR **Malaysia – No peatland**	**30 year emissions factor**	**30 year confidence interval**	**100 year emission factor**	**100 year confidence interval**
Forest	121 t CO_2e / ha	± 111 t CO_2e / ha	124 t CO_2e / ha	± 114 t CO_2e / ha
Grassland	3.5 t CO_2e / ha	± 15.1 t CO_2e / ha	-27.7 t CO_2e / ha	± 9.7 t CO_2e / ha
Mixed	33.0 t CO_2e / ha	± 32.3 t CO_2e / ha	1.8 t CO_2e / ha	± 29.7 t CO_2e / ha
Savannah	9.5 t CO_2e / ha	± 16.4 t CO_2e / ha	-21.6 t CO_2e / ha	± 9.8 t CO_2e / ha
Shrub	21.6 t CO_2e / ha	± 17.9 t CO_2e / ha	-9.5 t CO_2e / ha	± 10.7 t CO_2e / ha
Wetland	12.6 t CO_2e / ha	± 16.9 t CO_2e / ha	-18.6 t CO_2e / ha	± 9.9 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha

Table 74. Emissions from conversion of different land types in Malaysia to palm, assuming 5% of land expanded onto is peatland.
Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR **Malaysia – Low peatland**	**30 year emissions factor**	**30 year confidence interval**	**100 year emission factor**	**100 year confidence interval**
Forest	144 t CO_2e / ha	± 136 t CO_2e / ha	202 t CO_2e / ha	± 257 t CO_2e / ha
Grassland	27.0 t CO_2e / ha	± 65.5 t CO_2e / ha	50.8 t CO_2e / ha	± 362 t CO_2e / ha
Mixed	56.6 t CO_2e / ha	± 74.6 t CO_2e / ha	80.3 t CO_2e / ha	± 223 t CO_2e / ha
Savannah	33.1 t CO_2e / ha	± 65.5 t CO_2e / ha	56.8 t CO_2e / ha	± 220 t CO_2e / ha
Shrub	45.2 t CO_2e / ha	± 65.6 t CO_2e / ha	69.0 t CO_2e / ha	± 218 t CO_2e / ha
Wetland	36.1 t CO_2e / ha	± 65.5 t CO_2e / ha	59.9 t CO_2e / ha	± 221 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.00 g CO_2e / ha	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha

Table 75. Emissions from conversion of different land types in Malaysia to palm, assuming 33.3% of land expanded onto is peatland.
Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR **Malaysia – High peatland**	**30 year emissions factor**	**30 year confidence interval**	**100 year emission factor**	**100 year confidence interval**
Forest	275 t CO_2e / ha	± 111 t CO_2e / ha	637 t CO_2e / ha	± 95.2 t CO_2e / ha
Grassland	157 t CO_2e / ha	± 8.3 t CO_2e / ha	485 t CO_2e / ha	± 20.8 t CO_2e / ha
Mixed	187 t CO_2e / ha	± 28.6 t CO_2e / ha	515 t CO_2e / ha	± 30.4 t CO_2e / ha
Savannah	163 t CO_2e / ha	± 8.0 t CO_2e / ha	491 t CO_2e / ha	± 20.5 t CO_2e / ha
Shrub	176 t CO_2e / ha	± 7.7 t CO_2e / ha	504 t CO_2e / ha	± 20.0 t CO_2e / ha
Wetland	167 t CO_2e / ha	± 7.9 t CO_2e / ha	494 t CO_2e / ha	± 20.4 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha

Table 76. Emissions from conversion of different land types in Argentina to soybean. Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR Argentina	30 year emissions factor	30 year confidence interval	100 year emission factor	100 year confidence interval
Forest	61.0 t CO_2e / ha	± 81.8 t CO_2e / ha	61.0 t CO_2e / ha	± 81.7 t CO_2e / ha
Grassland	11.2 t CO_2e / ha	± 57.7 t CO_2e / ha	11.2 t CO_2e / ha	± 57.7 t CO_2e / ha
Mixed	22.6 t CO_2e / ha	± 46.8 t CO_2e / ha	22.6 t CO_2e / ha	± 46.8 t CO_2e / ha
Savannah	14.4 t CO_2e / ha	± 39.8 t CO_2e / ha	14.4 t CO_2e / ha	± 39.8 t CO_2e / ha
Shrub	20.6 t CO_2e / ha	± 41.2 t CO_2e / ha	20.6 t CO_2e / ha	± 41.2 t CO_2e / ha
Wetland	15.9 t CO_2e / ha	± 39.8 t CO_2e / ha	15.9 t CO_2e / ha	± 39.8 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha

Table 77. Emissions from conversion of different land types in Colombia to palm. Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR Colombia	30 year emissions factor	30 year confidence interval	100 year emission factor	100 year confidence interval
Forest	157 t CO_2e / ha	± 3.8 t CO_2e / ha	126 t CO_2e / ha	± 3.1 t CO_2e / ha
Grassland	3.4 t CO_2e / ha	± 15 t CO_2e / ha	-28 t CO_2e / ha	± 9.7 t CO_2e / ha
Mixed	42.0 t CO_2e / ha	± 15 t CO_2e / ha	10.7 t CO_2e / ha	± 10 t CO_2e / ha
Savannah	9.4 t CO_2e / ha	± 16 t CO_2e / ha	-22 t CO_2e / ha	± 9.8 t CO_2e / ha
Shrub	21.4 t CO_2e / ha	± 18 t CO_2e / ha	-10 t CO_2e / ha	± 11 t CO_2e / ha
Wetland	12.1 t CO_2e / ha	± 17 t CO_2e / ha	-19 t CO_2e / ha	± 9.9 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha	0.00 t CO_2e / ha	± 0.0 t CO_2e / ha

Table 78. Emissions from conversion of different land types in Brazil to soybean. Based on Winrock data for the US EPA (2010).

CONVERSION FACTOR Brazil	30 year emissions factor	30 year confidence interval	100 year emission factor	100 year confidence interval
Forest	131 t CO_2e / ha	± 108 t CO_2e / ha	165 t CO_2e / ha	± 108 t CO_2e / ha
Grassland	30.6 t CO_2e / ha	± 18.2 t CO_2e / ha	30.6 t CO_2e / ha	± 18.2 t CO_2e / ha
Mixed	57.6 t CO_2e / ha	± 30.8 t CO_2e / ha	57.6 t CO_2e / ha	± 30.8 t CO_2e / ha
Savannah	39.7 t CO_2e / ha	± 15.9 t CO_2e / ha	39.7 t CO_2e / ha	± 15.9 t CO_2e / ha
Shrub	58.9 t CO_2e / ha	± 14.1 t CO_2e / ha	58.9 t CO_2e / ha	± 14.1 t CO_2e / ha
Wetland	44.7 t CO_2e / ha	± 16.9 t CO_2e / ha	44.7 t CO_2e / ha	± 16.9 t CO_2e / ha
Barren	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha	0.00 t CO_2e / ha	± 0.00 t CO_2e / ha

Table 79. Emissions from reversion of cropland to different types of land in the EU. Based on Winrock data for the US EPA (2010).

REVERSION FACTOR EU	30 year emissions factor	30 year confidence interval	100 year emission factor	100 year confidence interval
Forest	-28.8 t CO_2e / ha	± 24 t CO_2e / ha	-88.9 t CO_2e / ha	± 106 t CO_2e / ha
Grassland	-20.9 t CO_2e / ha	± 25 t CO_2e / ha	-21.0 t CO_2e / ha	± 25.0 t CO_2e / ha
Mixed	-25.5 t CO_2e / ha	± 22 t CO_2e / ha	-25.5 t CO_2e / ha	± 22.0 t CO_2e / ha
Savannah	-34.7 t CO_2e / ha	± 23 t CO_2e / ha	-34.7 t CO_2e / ha	± 23.4 t CO_2e / ha
Shrub	-39.4 t CO_2e / ha	± 25 t CO_2e / ha	-39.4 t CO_2e / ha	± 25.3 t CO_2e / ha

Annex 3. Breakdown of the palm biodiesel ILUC factors by scenario

Table 80. ILUC factor breakdown for all scenarios of the palm biodiesel chain
Units: g CO_2e / MJ biofuel

Type of impact	Geographical location	Scen. 1	Scen. 2	Scen. 3	Scen. 4	Scen. 5	Scen. 6	Scen. 7	Scen. 8	Scen. 9	Scen. 10
Expansion of palm area	Indonesia	76.11	29.65	39.74	90.50	12.45	22.54	73.30	6.78	16.87	67.63
	Malaysia	61.28	23.76	31.28	72.87	11.89	19.40	60.99	5.87	13.38	54.97
	Colombia	0.85	1.01	1.01	1.01	1.01	1.01	1.01	0.16	0.16	0.16
Avoided soybean expansion	Argentina	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.07	0.07	0.07
	Brazil	0.50	0.50	0.50	0.50	0.50	0.50	0.50	0.17	0.17	0.17
Avoided coconut expansion	Indonesia	68.33	26.62	35.68	81.26	11.18	20.23	65.81	6.09	15.15	60.73
Avoided wheat expansion	EU	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.05	0.05	0.05
	Outside EU	1.39	1.39	1.39	1.39	1.39	1.39	1.39	0.56	0.56	0.56
Additional palm production to replace soybean oil	Indonesia	0.22	0.09	0.12	0.27	0.04	0.07	0.21	0.02	0.05	0.20
	Malaysia	0.22	0.09	0.11	0.26	0.04	0.07	0.22	0.02	0.05	0.20
Total ILUC factor		**68.18**	**25.80**	**34.40**	**81.48**	**12.08**	**20.68**	**67.75**	**5.92**	**14.52**	**61.59**

Annex 4. Breakdown of the oilseed rape biodiesel ILUC factors by scenario

Table 81. ILUC factor breakdown for all scenarios of the oilseed rape biodiesel chain
Unit: g CO_2e / MJ biofuel

Type of impact	Geographical location	Scen. 1	Scen. 2	Scen. 3	Scen. 4	Scen. 5	Scen. 6
Cereal area expansion	EU	33.88	33.88	33.88	33.88	33.88	19.28
	Ukraine	18.18	18.06	18.18	18.18	18.18	14.95
	Canada	-	-	-	-	-	23.17
Additional palm production to replace rapeseed oil		-	0.16	-	-	-	-
Feed wheat displacement	EU	-0.61	-0.61	-0.30	-0.61	-2.12	-0.38
	Canada	-	-	-	-	-	-2.53
Soybean displacement	Argentina	-11.27	-11.27	-5.62	-11.27	-6.98	-11.27
	Brazil	-29.21	-29.21	-14.58	-29.21	-18.17	-29.21
Additional palm production to replace soybean oil		6.38	6.38	3.19	3.83	4.03	6.38
Total ILUC factor		**17.36**	**17.39**	**34.74**	**14.81**	**28.82**	**20.39**

Annex 5. Breakdown of the soy biodiesel ILUC factors by scenario

Table 82. ILUC factor breakdown for all scenarios of the soybean biodiesel chain
Unit: g CO_2e / MJ biofuel

Type of impact	Geographical location	Scen. 1	Scen. 2	Scen. 3
Additional palm production for soybean oil replacement	China	62.17	4.52	48.32
Additional rapeseed oil production for soybean oil replacement	China	3.47	3.47	6.25
	Mexico	0.48	0.48	0.48
	Canada	0.25	0.25	0.25
Total ILUC factor		**66.38**	**8.73**	**55.31**

Annex 6. Breakdown of the wheat bioethanol ILUC factors by scenario

Table 83. ILUC factor breakdown for all scenarios of the wheat bioethanol chain
Unit: g CO_2e / MJ biofuel

Type of impact	Geographical location	Scen. 1	Scen. 2	Scen. 3	Scen, 4	Scen. 5	Scen. 6	Scen. 7	Scen. 8
Wheat area expansion	EU	14.00	15.34	47.18	16.67	16.67	16.67	16.67	16.67
	Australia	17.62							
	Canada	14.45							
	Ukraine		3.89						
Feed wheat displacement	EU	-2.92	-2.48	-1.21	-2.44	-1.21	-2.44	-2.44	-2.44
Soybean displacement	Argentina	-16.73	-16.73	-16.86	-16.73	-8.32	-20.78	-12.68	-16.73
	Brazil	-43.36	-43.36	-43.70	-43.36	-21.60	-55.48	-31.24	-43.36
Additional palm production for soybean oil replacement		9.47	9.47	9.47	9.47	4.74	9.47	9.47	5.69
Total ILUC factor		-7.47	-33.86	-5.11	-36.39	-9.73	-52.56	-20.21	-40.17

Annex 7. Breakdown of the sugarcane bioethanol ILUC factors by scenario

Table 84. ILUC factor breakdown for all scenarios of the sugarcane bioethanol chain
Unit: g CO_2e / MJ biofuel

Type of impact	Geographical location	Scen. 1	Scen. 2	Scen. 3	Scen. 4	Scen. 5	Scen. 6	Scen. 7	Scen. 8	Scen. 9	Scen. 10	Scen. 11	Scen. 12	Scen. 13	Scen. 14
Sugarcane area expansion	Brazil	10.31	10.98	6.29	11.21	6.96	7.53	13.69	3.31	8.22	14.39	2.28	16.42	18.87	14.50
	Argentina	-	-	1.71	-	-	-	-	-	-	-	-	-	-	-
	S. America	0.93	0.99	0.99	1.00	0.93	1.00	0.93	0.90	0.93	1.00	1.00	0.93	0.93	1.30
	Central America	4.81	4.81	4.81	3.28	4.81	3.28	4.81	4.81	4.81	3.28	3.28	4.81	4.81	5.92
	China	0.11	0.11	0.11	0.04	0.11	0.04	0.11	0.11	0.11	0.04	0.04	0.11	0.11	0.11
	Indonesia	1.00	1.00	1.00	0.43	1.00	0.43	1.00	1.00	1.00	0.43	0.43	1.00	1.00	1.86
	Philippines	0.45	0.45	0.45	0.19	0.45	0.19	0.45	0.45	0.45	0.19	0.19	0.45	0.45	0.84
	Southern Africa	0.04	0.01	0.01	0.02	0.04	0.02	0.04	0.04	0.04	0.02	0.02	0.04	0.04	0.07
	Thailand	0.99	0.99	0.99	0.42	0.99	0.42	0.99	0.99	0.99	0.42	0.42	0.99	0.99	1.36
	USA	0.17	-	-	0.15	0.17	0.15	0.17	0.17	0.17	0.15	0.15	0.17	0.17	0.27
Total ILUC factor		18.81	19.34	16.37	16.75	15.46	13.07	22.19	11.79	16.72	19.92	7.82	24.92	27.38	26.22

Annex 8. Pasture stock rate calculations for sugarcane ethanol chain

The numbers used in the calculations below are also based on data from the IBGE Agricultural Census and compiled for us by ICONE, unless otherwise stated.

Extrapolation of historical rates of increase in stock rate

The CAGR of stock rate in the Centre-South region from 1970 to 2006 was 2.2%, whereas in the North region, the CAGR was 3.2%. This leads to stock rate projections in the Centre-South in 2020 of 1.53 head / ha compared with 1.83 head / ha in North. This means that 1 ha pasture displaced in the Centre-South region leads to 0.84 ha pasture expansion in the North region.

Calculations:

Stock rate in Centre-South in 1970 = 0.52 heads / ha

Stock rate in Centre-South in 2006 = 1.13 heads / ha

CAGR 1970-2020 of stock rate in Centre-South = (1.13/0.52) ^ (1/(2006-1970)) – 1 = 2.2%

Stock rate in Centre-South in 2020 = 1.13 x (1 + 2.2%) ^ (2020-2006) = 1.53 heads / ha

Stock rate in North in 1970 = 0.39 heads / ha

Stock rate in North in 2006 = 1.18 heads / ha

CAGR 1970-2020 of stock rate in North = (1.18/0.39) ^ (1/(2006-1970)) – 1 = 3.2%

Stock rate in North in 2020 = 1.18 x (1 + 3.2%) ^ (2020-2006) = 1.83 heads / ha

Pasture equivalence = 1.53 / 1.83 = 0.84 ha expansion in North / ha displaced in Centre South

Above baseline pasture intensification in the North leading to no pasture expansion

For 1 ha of displaced pasture in the Centre-South region to lead to 0 ha of pasture expansion in the North region, the stock rate in the North would have to increase to 1.9 or 2.0 head / ha. This stock rate depends on assumptions such as the number of cattle heads in the North region in the baseline scenario and the number of cattle heads displaced from the Centre-South region. Annex 8 provides more details on the assumptions behind this calculation.

Calculations (example of scenario 8):

Cattle heads in Brazil in 2020 = 225 million heads (FAPRI (2010) projections)

Percentage of cattle heads in North out of whole of Brazil in 1970 = 2%

Percentage of cattle heads in North out of whole of Brazil in 2006 = 18%

CAGR of percentage of cattle heads in North = 6%

Percentage of cattle heads in North out of whole of Brazil in 2020 = 39%

Cattle heads in North in baseline projection in 2020 = 225 x 30% = 88 million heads

Stock rate in Centre-South in 2020 = 1.53 heads / ha

Pasture land displaced by sugarcane production in Centre-South in 2020 = 1.4 million ha

Cattle heads displaced from Centre-South to North = 1.4 x 1.53 = 2.2 million heads

Stock rate in North in baseline projection in 2020 = 1.83 heads / ha

Pasture land in North in baseline projection in 2020 = 88 x 1.83 = 48 million ha

Stock rate in North in biofuel projection in 2020 = (88 + 2.2) / 48 = 1.9 heads / ha

Annex 9. Overview of causal-descriptive model

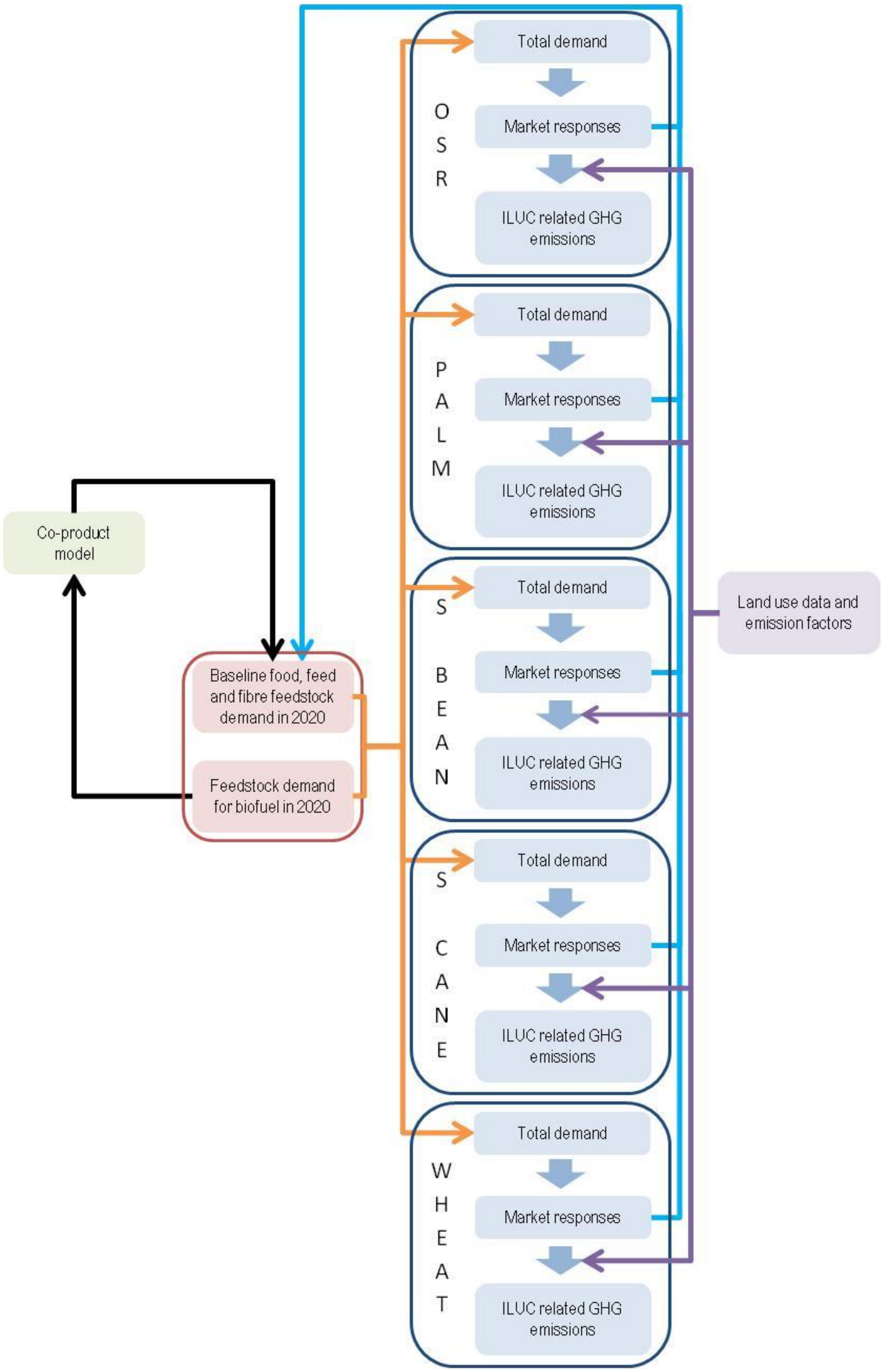

Figure 44. Data and information flows between the different modules of the causal-descriptive model developed for this study.